RRB-89-59

This manuscript has been prepared for

an IWD Scientific Series Report

and the contents are subject to change.

This copy is to provide information

prior to publication.

CONTAMINANT HYDROGEOLOGY OF TOXIC ORGANIC CHEMICALS AT A DISPOSAL SITE, GLOUCESTER, ONTARIO. 2. REMEDIAL INVESTIGATION

by
R.E. Jackson, S. Lesage, M.W. Priddle,
A.S. Crowe, and S. Shikaze

National Water Research Institute
867 Lakeshore Road
Burlington, Ontario, L7R 4A6

July 1989
NWRI Contribution #89-154

MANAGEMENT PERSPECTIVE

This report, written by members of the Ground Water Contamination Project, RRB, NWRI, presents (1) research results on the migration and fate of organic contaminants in a shallow ground water flow system; (2) a discussion of remedial options available to the site owner, Transport Canada, and (3) an example of a remedial investigation for use by Federal and Provincial agencies in the forthcoming Canadian hazardous waste site remediation programme. The work was funded by Transport Canada and NWRI.

The Special Waste Compound of the Gloucester landfill at Ottawa International Airport was used from 1972-80 by EPS/DOE to dispose Federal laboratory wastes. Part 1 of this report, published by NHRI in 1985, documented the substantial ground water contamination that had occurred beneath the site.

Ground water quality monitoring by NWRI during the period 1986-89 has shown that a plume of organic chemicals has migrated off site from the Gloucester landfill, near Ottawa, Ontario. The leading edge of the plume contains high concentrations of chloride and 1,4-dioxane, while most halogenated compounds remain directly below the site.

The remedial alternatives considered for the site are reviewed and preliminary remedial measures taken on behalf of Transport Canada are described. Containment systems, such as slurry walls, are believed unsuitable because of the permeable nature of the bedrock beneath the site. The principal remedial option recommended by

consultants to Transport Canada should be successful in decontaminating large parts of the aquifer, however, the aquitard underlying the SWC may be extensively contaminated by non-aqueous phase liquids and therefore require excavation. In-situ biorestoration may be required to complete aquifer decontamination following the 5-year pump and treat operation.

PERSPECTIVE-GESTION

Le présent rapport, rédigé par les membres du Projet de contamination des eaux souterraines, de la Direction de la recherche sur les cours d'eau, Institut national de recherche sur les eaux, contient 1) les résultats des recherches sur la migration et le devenir des contaminants organiques dans les eaux souterraines peu profondes, 2) une étude des différentes solutions possibles offertes au propriétaire du site d'enfouissement, Transport Canada, et 3) un exemple d'une étude des mesures correctives à l'intention des organismes fédéraux et provinciaux pour la mise en oeuvre prochaine du programme canadien de restauration des sites d'enfouissement de déchets dangereux. Les travaux ont été financés par Transport Canada et l'INRE.

L'enceinte de déchets spéciaux du site d'enfouissement de Gloucester à l'aéroport international d'Ottawa a été utilisée par le SPE/MDE pour évacuer les déchets des laboratoires fédéraux, de 1972 à 1980. La première partie du rapport, publiée par l'INRE en 1985, traite de la forte contamination des eaux souterraines sous-jacentes au site.

La surveillance de la qualité des eaux souterraines par l'INRE durant la période de 1986 à 1980 a révélé qu'un panache de produits chimiques organiques s'était éloigné du site d'enfouissement de Gloucester, près d'Ottawa, en Ontario. Le front avant du panache contient de fortes concentrations de chlorures et de 1,4-dioxane,

alors que presque tous les composés halogénés sont restés directement sous le site.

Les mesures correctives retenues sont examinées et les mesures préliminaires prises au nom de Transport Canada sont décrites. Les systèmes de confinement, comme les murs de boue, ne sont guère appropriés, étant donné la perméabilité du socle sous-jacent au site. La principale solution recommandée par les experts-conseils devrait permettre de décontaminer une bonne partie la nappe aquifère; toutefois, l'aquitard sous-jacent à l'enceinte pourrait être fortement contaminé par des liquides en phase non aqueuse, de sorte que des travaux d'excavation pourraient se révéler nécessaires. La biorestauration in situ sera peut-être nécessaire pour compléter la décontamination de la nappe aquifère, après les opérations de pompage et d'épuration échelonnées sur cinq ans.

ABSTRACT

Hydrogeochemical research by NWRI scientists during the period 1986-89 has shown that a large plume of organic chemicals has continued to migrate through an outwash aquifer following disposal at the Special Waste Compound (SWC) at the Gloucester landfill, near Ottawa, Ontario. The leading edge of the plume, characterized by high concentrations of chloride and aliphatic ethers, e.g., 1,4-dioxane, has moved off site. Most of the halogenated hydrocarbons have not been transported so far and remain directly below the SWC in the outwash aquifer and the overlying aquitard. In this zone there is considerable evidence of dehalogenation reactions, which are probably microbially catalyzed.

Field and laboratory experiments were undertaken to determine the mobility of the aliphatic ethers and some halogenated hydrocarbons in the outwash aquifer. The individual retardation factors derived from these experiments show good correlation with the octanol-water coefficient of each compound. Comparison of the lab and field experiments indicates that laboratory soil columns yield reasonable estimates of the retardation factors determined in the field.

The remedial alternatives considered for the site are reviewed and preliminary remedial measures taken on behalf of the site owner, Transport Canada, are described. Containment systems, such as slurry walls, are unsuitable because of the permeable nature of the bedrock

beneath the SWC. The principal remedial option recommended by consultants to Transport Canada should be successful in decontaminating large parts of the aquifer, however, the aquitard underlying the SWC may be extensively contaminated by non-aqueous phase liquids and therefore require excavation. In-situ biorestoration may be required to complete aquifer decontamination following the 5-year pump and treat operation.

RÉSUMÉ

Les recherches hydrogéochimiques menées par les chercheurs de l'INRE entre 1986 et 1989 ont révélé qu'un vaste panache de produits chimiques organiques a poursuivi sa migration dans une nappe aquifère d'alluvions glaciaires, sous une enceinte de déchets spéciaux installée au site d'enfouissement de Gloucester, près d'Ottawa, en Ontario. Le front avant du panache, caractérisé par de fortes concentrations de chlorures et d'éthers aliphatiques, par ex. 1,4-dioxane, s'est éloigné du site. La plupart des hydrocarbures halogénés n'ont été migré si loin, et sont resté directement sous l'enceinte dans nappe aquifère et l'aquitard sus-jacent. Cette zone renferme de nombreuses traces de réactions de déshalogénation qui sont probablement catalysées par des microorganismes.

Des expériences sur le terrain et en laboratoire ont été entreprises afin d'évaluer la mobilité des éthers aliphatiques et de quelques hydrocarbures halogénés dans la nappe aquifère. Les coefficients de retard individuels calculés au cours de ces expériences montrent une bonne corrélation avec le coefficient octanol-eau de chaque composé. La comparaison entre les expériences en laboratoire et sur le terrain indique que les colonnes de sol analysées en laboratoire fournissent des estimations raisonnables des coefficients de retard obtenus sur le terrain.

Les différentes solutions possibles pour restaurer la nappe aquifère sont examinées et les mesures préliminaires prises au nom du

propriétaire du site, soit Transport Canada, sont décrites. L'utilisation de systèmes de confinement, comme les murs de boue, n'est guère appropriée, vu la perméabilité du socle sous-jacent à l'enceinte. La solution recommandée par les experts-conseils devrait permettre de décontaminer une bonne partie la nappe aquifère; toutefois, l'aquitard sous-jacent à l'enceinte pourrait être largement contaminé par les liquides en phase non aqueuse, de sorte que des travaux d'excavation pourraient se révéler nécessaires. La biorestauration sur place sera peut-être requise pour compléter la décontamination de la nappe aquifère après cinq ans de pompage et d'épuration.

TABLE OF CONTENTS
Part 2 - Remedial Investigation

Chapter 1: INTRODUCTION
Ground Water Quality at Hazardous Waste Sites in Eastern Canada
Health Risks of Contaminated Ground Water
The Need for Reliable Remedial Investigations

Chapter 2: THE GLOUCESTER LANDFILL AND OUTWASH AQUIFER
Site Description
Hydrogeology
Inorganic Ground Water Quality
Organic Contaminant Migration

Chapter 3: ANALYSIS AND FATE OF ORGANIC PRIORITY POLLUTANTS IN THE OUTWASH AQUIFER
Sampling and Analysis of Organic Contaminants in Ground Water
Results of 1988 Sampling Survey
Discussion of Contaminant Transport Mechanisms

Chapter 4: MOBILITY OF ORGANIC PRIORITY POLLUTANTS IN THE OUTWASH AQUIFER
Purge Well Test
Laboratory Column Tests

Chapter 5: DISCUSSION OF REMEDIAL OPTIONS

Chapter 6: SUMMARY AND CONCLUSIONS

Appendix 1: COMPLETE 1984 AND 1988 DATA SETS

LIST OF FIGURES

1. Location of five hazardous waste sites in Eastern Canada.
2. LNAPL migration patterns (after Schwille, 1975).
3. DNAPL migration patterns (after Feenstra and Cherry, 1988).
4. Range of carcinogenic potency in laboratory animals (after Gold, et al., 1984).
5. Hydrogeological cross section through surficial and bedrock aquifers, Ville Mercier, Quebec (from original by Ministère du l'Environnement Québec).
6. Location of study area near Ottawa and the locations of piezometers and multilevels in the monitoring well network. The property boundary of the site is identified by the railroad tracks running northwestwards across the figure.
7. Hydraulic head contours in meters above mean sea level (m.a.s.l.) along the cross section identified in the inset as measured during October, 1986. Squiggley arrow beneath the SWC indicates the potential for leakage through confining unit D. Note vertical exaggeration.
8. Hydraulic head contours in meters above mean sea level (m.a.s.l.) within the outwash aquifer as measured during October, 1986.
9. Concentration contours (in mg/L) of the maximum values of chloride from multilevel samplers within the outwash aquifer, (a) November, 1977, (b) July, 1983 and (c) May, 1988. The SWC is outlined at left.

10. Concentration contours (in mg/L) of the chloride plume along the indicated cross section (see inset), May 1988.
11. Plan view of three plumes of volatile organic chemicals in 1982 illustrating chromatographic dispersion (from Patterson et al., 1985).
12. Types of monitoring instruments: (a) the bundle-type multilevel and (b) the 5 cm i.d. piezometer with a dedicated submersible pump.
13. Flow chart of sampling and analytical methods.
14. Components of GC-MS system (from Swallow et al., 1988).
15. Total ion current chromatograms for multilevel sampling point 67M-11 at Gloucester.
16. Vertical location of multilevel inlet tubes and monitoring well screens along the cross section identified in the inset. As shown in the inset, some wells off the cross section are projected onto the line of section so as to indicate their well depth.
17. Congener profiles of chlorinated dibenzo-dioxins (CDD) and -furans (CDF) in a sample of ground water from 37M-19. The prefixes are as follows: T = tetra, Pn = penta, Hx = hexa, Hp = hepta.
18. Concentration contours (in µg/L) of a) benzene, b) chloroform, c) 1,1,1-trichloroethane, d) trichloroethene and e) 1,1-dichloroethene in the immediate proximity of the SWC, May 1988.
19. Concentration contours (in µg/L) of a) benzene, b) chloroform, c) 1,1,1-trichloroethane, d) trichloroethene and e) 1,1-dichloroethene along the cross section shown in Figure 17, May 1988.

20. Proposed anoxic biotransformation pathways for a) 1,1,1-trichloroethane or TCA under methanogenic conditions (Vogel and McCarty, 1987); b) tetrachloroethene or PCE, (Barrio-Lage et al., 1986); and c) 1,1,2-trichloro-1,2,2-trifluoroethane or F113 (Lesage et al., 1989). See text for other abbreviations. RD = Reductive dehalogenation reaction. E = Elimination or dehydrodehalogenation reaction.

21. Vertical concentration profiles (in µg/L) of four organic solvent residues at monitor 16M during 1980, 1983 and 1988.

22. Vertical concentration profiles (in µg/L) of four organic solvent residues at monitor 37M during 1982, 1984 and 1988.

23. Time series data from two multilevel monitoring points within (37M) or adjacent to (16M) the SWC showing that concentrations of volatile organic chemicals have always been a small percentage of the solubility of any compound present in the ground water.

24. Rate of reduction of contaminant concentrations (diethyl ether or DEE, tetrahydrofuran or THF and 1,4-dioxane) during the purge well test.

25. Exploded view of laboratory soil column.

26. Simplfied diagram of complete column apparatus.

27. Breakthrough curves of iodide (V = 45 cm/day) and diethyl ether showing both measured and fitted values using the LEA model (iodide) and the kinetic model (diethyl ether).

28. Breakthrough curves of 1,2-dichloroethane, chloroform or tetrachloromethane, 1,1-dichloroethene and benzene for the same test as Figure 27.

29. Geometry of the purge and recharge well network as recommended by GEC (1985b).

30. Cross-sectional profile of benzene contamination in 1984 and 1988.

31. Cross-sectional profile of chloroform contamination in 1984 and 1988.

32. Cross-sectional profile of 1,1,1-trichloroethane contamination in 1984 and 1988.

33. Cross-sectional profile of trichloroethene contamination in 1984 and 1988.

34. Estimation of the number of pore volumes that must be purged from the outwash aquifer to achieve 90% decontamination. Pore volume values for diethyl ether, tetrahydrofuran and 1,4-dioxane only are obtained from Figure 24. Other values are extrapolated.

LIST OF TABLES

1. Information concerning five hazardous waste sites in Eastern Canada.
2. Concentration ranges of contaminants of concern in ground waters at five hazardous waste sites in eastern Canada.
3. Drinking water quality guidelines values in µg/L (ppb).
4. Ground water chemistry of selected toxic organic chemicals in µg/L at two of the purge wells, Mercier, Quebec, in August 1988, compared to U.S. EPA maximum contaminant levels in drinking water.
5. Inorganic ground water quality data at selected multilevel samplers (16M, 37M) and piezometers (41P) within the outwash aquifer.
6. E_H and dissolved sulfide in ground waters from the outwash aquifer directly beneath the SWC.
7. pH, E_H and dissolved oxygen in ground waters (monitor 37M-9) beneath the SWC in 1982, 1983 and 1988.
8. Volatile organics present in the outwash aquifer, May 1988.
9. Volatile organic compounds tentatively identified by computerized library searches.
10. Volatile organic compounds (in µg/L) identified in monitoring well network, May 1988.
11. Product distribution expressed as percentage of total chlorinated ethanes/ethenes found in 37M, 67M and 92M.

12. Retardation factors.

13. Corrective action alternatives.

14. Volatile organic compounds measured by GC/MS in µg/L in ground waters from selected multilevel samplers within the outwash aquifer.

FREQUENTLY USED ABBREVIATIONS IN THE TEXT

DCA: dichloroethane

DCE: dichloroethene

DEE: ciethyl ether

DNAPL: dense, non-aqueous phase liquid

EPA: U.S. Environmental Protection Agency

Freon 113: 1,1,2-trichloro-1,2,2-trifluoroethane

GAO: U.S. General Accounting Office

GEC: A.J. Graham Engineering Consultants

GC-MS: gas chromatography/mass spectrometry

K_{ow}: octanol-water partition coefficient

LNAPL: light, non-aqeous phase liquid

M: multilevel, as in 54M

MSD: mass selective detector

P: piezometer, as in 134P

PAH: polyaromatic hydrocarbon

PCE: perchloroethylene or tetrachloroethene

QA: quality assurance

QC: quality control

R_f: retardation factor

RI/FS: remedial investigation/feasibility study

SIM: selected ion monitoring

SWC: the special waste compound at the Gloucester Landfill

TCA: 1,1,1-trichloroethane

TCDD: tetrachlorodibenzodioxin

TCE: tetrachloroethene

TCM: trichloromethane or chloroform

THF: tetrahydrofuran

VC: vinyl chloride or chloroethene

CONTAMINANT HYDROGEOLOGY OF TOXIC ORGANIC CHEMICALS AT A DISPOSAL SITE, GLOUCESTER, ONTARIO. 2. REMEDIAL INVESTIGATION

by

R.E. Jackson, S. Lesage, M.W. Priddle, A.S. Crowe and S. Shikaze

C H A P T E R 1

INTRODUCTION

Public concern over the health risks associated with hazardous waste sites and the consumption of ground waters possibly polluted by these sites has become acute in the past decade. The assessment and remediation of such sites has involved the expenditure of millions of dollars of public and private funds in Canada and many times that amount in the U.S.A. With the announcement of the recent Federal/ Provincial agreement (April, 1989) to create a 5-year, $250 million hazardous waste site clean-up programme for orphan sites throughout Canada, it is essential that reliable and complete hydrogeologic remedial investigations be undertaken if the clean-up programme is to avoid many of the inefficiencies that have plagued the $5 billion Superfund of the U.S. EPA (e.g., Freeze and Cherry, 1989; Hirschhorn and Oldenburg, 1988). This report is an example of the kind and detail of hydrogeologic and hydrogeochemical information that will be required at Canadian hazardous waste sites if effective and efficient remedial technologies are to be chosen.

The purpose of this report is first to review and update the hydrogeological and hydrogeochemical data from the Gloucester landfill, near Ottawa, Ontario, and second to assess the remedial options for the restoration of the outwash aquifer beneath the Special Waste Compound. The extent of ground water contamination at this site has previously been documented in Jackson et al. (1985), i.e., part 1 of this report. In reviewing the hydrogeochemical data and remedial options, reference will be made to the reports of the various consultants, notably Water and Earth Sciences Ltd. (WESA) and Intera Technologies Ltd., who have undertaken hydrogeological studies either for the owner of the property, Transport Canada, or for Environment Canada. However, this report is primarily a presentation and interpretation of data from 1988 collected by the NWRI group associated with the continued assessment of ground water contamination arising from disposals of organic liquid wastes within the Special Waste Compound at the Gloucester landfill.

The remainder of this first Chapter is concerned with the origin and evolution of ground water contamination at hazardous waste sites, with the identity of the contaminants of concern at other sites in eastern Canada, with the health risks associated with the consumption of such contaminants in ground waters and with the need for a reliable, remedial investigation of the hydrogeology of a site.

Chapter 2 contains a review of waste disposal, hydrogeological and hydrogeochemical data from the Gloucester landfill from previous reports that are not now generally available. New hydrogeochemical

data collected by NWRI during 1988 are presented in Chapter 3 together with a discussion of the sampling and analytical methods used. This new data when coupled with the information on contaminant mobility presented in Chapter 4 allows the hydrogeologist to assess the appropriateness of the various remedial options (Chapter 5).

GROUND WATER QUALITY AT HAZARDOUS WASTE SITES IN EASTERN CANADA

It is becoming increasingly apparent that ground water quality at most hazardous waste sites is dominated by the occurrence of light and/or dense (relative to water) non-aqueous phase liquids, i.e., LNAPLs or DNAPLs. While it is entirely possible that sites contain neither NAPL, i.e., the source of all contaminants is an aqueous solution, the potential presence of NAPLs requires that this possibility be investigated prior to any remedial option being chosen.

Figure 1 shows the location of five hazardous waste sites in eastern Canada that have been investigated by the NWRI ground water contamination group. DNAPLs have been identified in ground water samples and by fouled downhole equipment at three of the five hazardous waste sites listed in Table 1; their presence in the other two sites is suspected but not confirmed. In addition to DNAPLs, samples from monitoring wells at Sydney, Nova Scotia contained LNAPL liquids.

Ground waters migrating around NAPL pools or through parts of aquifers containing residual NAPL emulsions, i.e., droplets or

"ganglia", will become contaminated by mass transfer of the NAPLS to the ground water. As Hunt et al. (1988a) have shown, such mass transfer is very slow and dependent upon the solubility and molecular diffusivity of the organic compounds comprising the NAPL, the rate of ground water flow past the NAPL zones and the surface area of the NAPL. The processes by which sinking LNAPL and DNAPL plumes dissolve and contaminate a whole aquifer is shown in Figures 2 and 3.

The concentration of the "i"th component of the NAPL in ground water is given by the approximation (Stumm and Morgan, 1981; Banerjee, 1984):

$$C_i = x_i C_s$$

where x_i is the mole fraction of the "i"th component in the DNAPL and C_s is the aqueous solubility of the component. Two important observations may be made about this expression.

First, the equilibrium concentration of the "i"th contaminant in ground water adjacent to a NAPL pool will be less than the aqueous solubility limit by a factor proportional to the mole fraction of the contaminant in the pool. This is because the contaminant preferentially partitions into the pool (Reinhard et al., 1984). Second, in the case of several contaminants being present in a NAPL pool with similar mole fractions, the concentration of each in adjacent ground water will be a function of their respective aqueous solubilities. With time, the more soluble components will preferentially dissolve in

the ground water because such mass transfer is primarily a function of aqueous solubility rather than ground water flow rate (Anderson et al., 1987). This phenomenon is observed in the weathering of oil seeps (Hunt, 1979) and in the remediation of solvent plumes in ground water (Feenstra, S., personal communication).

Volatile halogenated aliphatics, pesticides and aromatic hydrocarbons, have dissolved in ground water from NAPL pools and/or ganglia at most of the sites listed in Table 1 and have created plumes with aqueous concentrations of tens of mg/L, as shown in Table 2. Because these concentrations exceed drinking water limits by as much as one thousand fold, the presence of these contaminants is of foremost concern.

HEALTH RISKS OF CONTAMINATED GROUND WATER

Guideline values for Canadian drinking water quality are set by a Federal-Provincial subcommittee. Some of the more relevant values are shown in Table 3 along with comparable U.S. EPA values. In most cases, Canadian values do not exist for those compounds identified in the plumes migrating from the hazardous waste sites listed in Table 1, consequently U.S. EPA values are frequently adopted by Canadian regulatory authorities.

The setting of these values is done in an extremely conservative way involving the extrapolation of animal cancer tests and, sometimes, human epidemiological data, to predict human health risks. Information from the most detailed assessment of these risks yet published

(Gold et al., 1984) is displayed in schematic form in Figure 4, which includes a carcinogenic rating of a wide variety of compounds most of which have been identified in the five hazardous waste sites considered in this paper. While this procedure has been criticized by leading toxicologists as being too conservative (Ames et al., 1987; Smith, 1988), it is, nevertheless, a methodology that is widely used by public health agencies. Thus, the health risk estimation technique forces chemists and hydrogeologists to design sampling and analysis programmes to identify and quantitate potential carcinogens in ground water at concentrations which comply with the guideline values, the so-called "compliance" levels.

The risk associated with drinking ground water contaminated with any of the compounds shown in Figure 4 is assumed proportional to the value of TD_{50} - "the dose rate (in mg/kg body weight/day) which, if administered chronically for the standard lifetime of the species (e.g., male rate), will halve the probability of remaining tumorless throughout that period" (Gold et al., 1984). This index of carcinogenic potency ranges from the most toxic - an isomer of tetrachlorodibenzodioxin (2,3,7,8-TCDD) - to the least - green food dye no. 1. However, the contaminants of greatest concern to the hydrogeologist are those that are not only toxic, but also mobile and persistent in the subsurface (Jackson et al., 1985).

Consequently, TCDD isomers, which are occasionally found in soils and ground water, may be of less concern at a particular hazardous waste site because of their very low mobility due to their low solubility. Most of the organic contaminants of concern identified in

Table 3 occupy that region in Figure 4 between 1 mg/kg/day and 1 g/kg/day. That compound with the lowest drinking water guideline value in Table 2 is vinyl chloride or chloroethene, which is a known human carcinogen and which is produced in ground water by the biodegradation of several common chlorinated organic solvents (Wolf et al., 1987). Because of its combined toxicity, mobility and persistence, vinyl chloride is the contaminant of greatest concern at many sites, even though these sites may contain compounds identified in Figure 4 as being of much greater carcinogenic potential. This fact necessitates the undertaking of remedial investigations which are optimized for such contaminants of concern.

THE NEED FOR RELIABLE REMEDIAL INVESTIGATIONS

The assessment of hydrogeological and hydrogeochemical conditions at a hazardous waste site should lead to a realistic appraisal of the potential of the various remedial technologies to achieve the required corrective action. In American parlance these two steps are known as the remedial investigation and feasibility study or RI/FS when they are issued as a single report. Because of the inadequacy of many remedial investigations in the U.S.A. (GAO, 1988) and Canada, the prescribed remedial technologies have failed to attain their stated objectives, which may have been inattainable in the first place.

An excellent example of the failure of a remedial investigation to identify the actual conditions at a hazardous waste site and

thereby undermine the objectives of the feasibility study is that of Ville Mercier, Quebec (Martel, 1988). During 1968/72, approximately 40,000 m^3 of liquid industrial wastes were disposed in lagoons in an abandoned gravel pit near Ville Mercier on the south shore of the St. Lawrence River, 20 km from Montreal (see Figure 1). By 1971, local wells were closed due to contamination and by 1972, disposals were ordered ended. However, by May 1988, a plume of volatile organic chemicals had migrated at least 4.8 km from the disposal areas and had caused the closure of industrial and municipal water supply wells in Ste-Martine (see Figure 5) as well as numerous domestic wells between Ste-Martine and Mercier.

In 1982, an RI/FS was undertaken that lead to the commissioning of a treatment plant in June 1984 to decontaminate the ground water withdrawn from the aquifer beneath the former lagoons. It was intended that the treated water would have "similar characteristics as drinking water" and could be used by Mercier for its potable water supply after five years of pump and treat operations (Simard and Lanctot, 1987). This treatment plant and three purge wells operated from June 1984 to June 1985 at 1.2 million m^3/yr at a cost of $7.4 million but failed to produce any significant improvement in the ground water quality entering the treatment plant vis-a-vis the drinking water guidelines. As of August 1988, concentrations of four toxic organic chemicals in the purge wells adjacent to the lagoons were still substantially in excess of drinking water guidelines (Table 4).

The failure of the pump and treat operations to attain the objective of producing drinking water for Ville Mercier after five years of operation was due to the presence of DNAPL pools within the sand and gravel surficial aquifer and the fractures of the bedrock. Chemical analysis of one of these pools by Laval University indicated that it was composed of a complex mixture of refined petroleum hydrocarbons (54%), volatile organic solvents (18%) and PAHs (4.7%) (H. Pakdel, Chemical Engineering Department, Laval University, personal communication). Ground water in contact with this DNAPL contained volatile organic solvents in concentrations of only 1 or 2% of their aqueous solubilities, indicating relatively slow mass transfer between the pool and nearby ground water. Consequently, aquifer purging operations failed to remove this subsurface source of contamination from beneath the lagoons, with the exception of a pool that was pumped into the treatment plant and fouled the air stripping columns, clarifier unit and granular activated carbon columns.

Because the DNAPLs were not identified in the 1982 RI/FS, the remedial technology, pump and treat, chosen to decontaminate the aquifer could only contain the migration of the dissolved contaminants from the DNAPL but could not remove the DNAPL pools from the aquifer (Martel, 1988). This conclusion has also been reached in the U.S.A. in similar situations (Feenstra and Cherry, 1988; Mackay and Cherry, 1989). Apart from excavation of shallow DNAPL pools, there is no proven remedial technology to remove DNAPLs, although some techniques appear to show great promise (see Chapter 5).

The RI/FS of the Mercier site was completed for $300,000 (Simard and Lanctot, 1987), of which less than $100,000 was for hydrogeological and hydrogeochemical investigation. At the Gloucester landfill where only 100 m^3 of liquid wastes were disposed, Transport Canada has spent approximately $450,000 on hydrogeological and hydrogeochemical studies and the research institutes of Environment Canada (NHRI, NWRI, RRETC, WTC) have spent at least as much. In no sense then can the 1982 Mercier RI/FS be considered to have been reliable and complete because of the very small budget allocated to the hydrogeological investigation given the scale of disposals at the site.

Three items were particularly absent from the original Mercier RI/FS. Firstly, the sand and gravel aquifer beneath the lagoons was not sufficiently well investigated and therefore the DNAPL pools were not identified. Had they been identified, it would have been possible to determine, by laboratory column experiments, that their removal would be impossible by pump and treat operations. In fairness, it must be added that the DNAPL problem was not fully appreciated by the hydrogeological community in 1982, however, the use of laboratory experiments in contaminant hydrogeology was well established. Secondly, no detailed assessment was conducted of the extent of contamination of the bedrock beneath the sand and gravel. Because of the very low bedrock porosities and high hydraulic gradients, ground water flow and contaminant transport is extremely rapid in the bedrock fractures and has lead to the development of the long plume extending towards Ste-Martine. Thirdly, no mathematical simulation modelling of

ground water flow and contaminant transport was undertaken to optimize the remedial operations. Mathematical modelling of the site (D. Isobel, Geological Engineering Department, Laval University, personal communication) has now shown that containment of the plume can be accomplished at much lower pumping rates than have been used over the period 1984/89, and therefore containment can be accomplished much more inexpensively due to lower granular activated carbon costs.

The Mercier remediation has been a costly learning experience that should be fully appreciated by Canadian regulators, engineers and hydrogeologists. While it is clear that not every site can be subject to the same extensive investigation that has characterized the Gloucester site, there must be some balance struck between funds allocated for a particular RI/FS, the health risks posed by the site and the value of the ground water resource that is threatened. However, it stands to reason that millions should not be poured into site remediation before a reliable and complete remedial hydrogeologic and hydrogeochemical investigation is performed.

CHAPTER 2

THE GLOUCESTER LANDFILL AND OUTWASH AQUIFER

Much hydrogeological and hydrogeochemical work has been undertaken at the Gloucester site during the past decade by both consultants to Transport Canada and research scientists and engineers of Environment Canada (e.g., Jackson et al., 1985). This chapter summarizes the hydrogeological and hydrogeochemical results arising from the site assessment studies of NHRI and these consultants from 1976 to 1985.

SITE DESCRIPTION

Between 1969 and 1980, the Government of Canada disposed of hazardous wastes, principally organic chemicals from its laboratories in Ottawa, at the nearby Gloucester landfill (Figure 6). The disposal operations took place within a "Special Waste Compound", hereafter referred to as the SWC, located at the southwest corner of the landfill. These operations involved the excavation of trenches (12 m long x 3 m x 3 m) into which bottles of liquid chemical wastes were placed and subsequently combusted by detonation of explosive charges set within them. The wastes consisted of non-chlorinated solvents (56.4 m^3), unspecified wood preservatives (30 m^3), chlorinated solvents (8.6 m^3) and smaller amounts of acidic and ferric chloride and other wastes (Graham Engineering Consultants, 1985a).

HYDROGEOLOGY

Detailed soil coring and hydrogeological testing have resulted in the identification of the five hydrostratigraphic units shown in Figure 7. Unit A, the fractured limestone bedrock, is encountered at depths between 25 and 30 m and is partly covered by a till (Unit B) of relatively low hydraulic conductivity. Unit C, overlying the till and/or bedrock, is a thick (up to 25 m) sequence of silts, sands, gravel and boulders known as glacial outwash and forms a semi-confined aquifer beneath the landfill. This outwash is overlain by a discontinuous layer of silt, unit D. The surficial aquifer, Unit E, may be up to 10 m thick and its composed of sands and gravels.

The outwash aquifer, Unit C, is composed mineralogically of feldspar (50+%), quartz (20%) and minor amounts of mica, calcite, dolomite and hornblende. The organic carbon content of the aquifer sediments averages 0.06%. The aquifer has a mean hydraulic conductivity of 10^{-4} m/s, a porosity of 0.30-0.35, and an average linear ground water flow velocity of 0.05 m/day (Jackson et al., 1985; Michel et al., 1984). The ground water temperature is in the range 9-10°C.

The ground water flow pattern beneath the SWC in October 1986 is shown in Figure 7. Downward ground water flow occurs from the surficial aquifer, Unit E, to the confined outwash aquifer. Some seepage appears to occur through permeable windows in the silty confining layer, Unit D. Once solutes enter the outwash aquifer, the vertical

equipotential lines dictate that they are transported horizontally to the east. Field tests have demonstrated that the underlying bedrock is hydraulically connected to the outwash aquifer (Jackson et al., 1985). Figure 8 shows the flow pattern in the outwash aquifer at the same time but in plan view.

INORGANIC GROUND WATER QUALITY

The distribution of chloride in plan and cross-section in the outwash aquifer is shown in Figures 9 and 10, respectively. The first samples collected at the Gloucester landfill in November 1977, showed values less than 5 mg Cl^-/L in that area (see Figure 9) where concentrations are now in excess of 250 mg Cl^-/L. Background chloride concentrations in ground waters from upgradient areas west of the SWC (e.g., 41P, see Figure 6) are less than 10 mg/L. Much higher levels occur beneath and downgradient from the SWC. The vertical configuration of the chloride plume (Figure 10) along a section extending downgradient from the SWC indicates that chloride is transported downward from the trenches within the SWC into the outwash aquifer and then downgradient to the east. The leading edge of this plume has crossed beneath the railroad tracks near Delzotto Avenue (Figure 9). This pattern of migration is consistent with the ground water flow directions shown in Figures 7 and 8. Based on ground water velocity measurements in the outwash aquifer and assuming no retardation, the plume is advancing at a rate of approximately 5 cm/day.

The pH of shallow ground waters upgradient of the waste site is on the order of 8.0, as shown in Table 5. Since rainfall and snowmelt are generally acidic (pH < 5), there is considerable neutralization occurring during infiltration and percolation as a result of the dissolution of carbonate minerals which are a significant component of the aquifer materials. Saturation index data (Table 5) indicate that all ground waters are near or above saturation with respect to calcite and ferrihydrite ($Fe(OH)_3$). Within the chloride plume the pH is generally less than 8.0 and occasionally less than 7.0. This is probably due to the hydrolysis of CO_2 evolved during the oxidation of the organic residues from the SWC, which produces titration alkalinities in excess of 10 mequiv/L compared with 3-4 mequiv/L in uncontaminated ground waters (Jackson et al., 1985).

Because of its importance in the transformation of organic compounds, the redox state of the ground water flow system was determined in detail, and has been monitored over an eight year period. Uncontaminated ground waters upgradient of the SWC (e.g., 41P) contain measurable quantities of dissolved oxygen (2-6 mg/L) and exhibit E_H values greater than 0.5 V. Within the chloride plume, ground waters have nondetectable quantities of dissolved oxygen (i.e., <0.3 mg/L) and E_H values ranging from -0.04 to 0.2 V; such values have been previously associated with sulfate reducing environments in ground water flow systems (Champ et al., 1979; Jackson and Patterson, 1982). ground water sulfide data for the outwash aquifer measured in 1983 from two multilevel monitors (37M and 67M) are shown in Table 6.

Because the highest levels of organic contamination of the outwash aquifer occurred in the period 1982-84, prior to some pumping at well 36W, it is important to note that no significant change in the pH or redox environment has occurred during the period 1982-88 (see Table 7).

ORGANIC CONTAMINANT MIGRATION

In addition to chloride contamination of the outwash aquifer, Jackson et al. (1985) reported substantial organic contamination. The patterns of organic contaminant migration in the outwash aquifer in 1982 resulting from disposals in the SWC suggest that the organic solvents have undergone chromatographic dispersion, i.e., aqueous-phase solute transport concurrent with sorption of the contaminants, principally by organic carbon particles within the aquifer sediments (Patterson et al., 1985). As shown in Figure 11, the organic contaminants have been transported distances that are inversely proportional to their respective octanol-water partition coefficients. Because the octanol-water partition coefficient (K_{ow}) provides a measure of a compound's propensity to become sorbed by organic matter (i.e., hydrophobic partitioning), it follows that the observed transport of the organic solvents by ground water is in accord with the hydrophobic sorption theory of Schwarzenbach and Westall (1981) and others. This pattern of behaviour has implications for aquifer decontamination. Those contaminants that have been transported furthest, that is the

most mobile, will be most easily removed from the aquifer by purging. The remedial system, therefore, must be designed to recover a variety of contaminants with widely varying mobilities.

CHAPTER 3

ANALYSIS AND FATE OF ORGANIC PRIORITY POLLUTANTS IN THE OUTWASH AQUIFER

The 1982 survey of the Gloucester site by NHRI indicated that there was extensive chloride and organic solvent contaminant migration towards the property boundaries as shown in Figures 9, 10 and 11. Although some heavy metal salts were disposed in the SWC, detailed hydrogeochemical sampling indicated that they were not present in the ground water in significant amounts (Jackson et al., 1985). Consequently, the contaminants of concern at the site have been the E.P.A. list of organic priority pollutants, a group of 129 toxic chemicals identified by Keith and Telliard (1979), some of which are shown in Figure 4 and known also to be carcinogenic. Therefore, detailed annual, organic pollutant surveys were conducted in 1984, 1987, 1988 and 1989. The results for 1984 and 1988 are tabulated in Appendix 1.

This chapter presents the general principles followed by NWRI in the sampling and analysis of organic priority pollutants and a discussion of the hydrogeological and hydrogeochemical results of the 1988 survey conducted by NWRI.

SAMPLING AND ANALYSIS OF ORGANIC CONTAMINANTS IN GROUND WATER

Sampling Methods

The sampling of organically-polluted ground waters requires a careful choice of monitoring instruments. The principal criterion of choice being that individual hydrostratigraphic units within a ground-water flow system must be individually sampled. Wells that penetrate more than a single unit provide little useful information (Grisak et al., 1978; Reilly et al., 1989). This is partly because the sample dilution that occurs in fully penetrating wells implies dispersion greater than actually takes place and also because this integrated sample may well indicate contaminant concentrations within the acceptable guidelines, although these guidelines may be exceeded within a particular unit.

Figure 12 shows two commonly used devices for sampling ground water quality - the bundle-type multilevel sampler and the 5 cm i.d. piezometer. Both provide the capability of sampling small zones of potentially-contaminated ground water. The first is frequently used to map the outlines of contaminant plumes in three-dimensional detail (e.g., Jackson et al., 1985). The second is generally used for monitoring ground-water quality where it is necessary to establish that this quality is in compliance with regulated or guideline values, consequently the well screen is made of an inert material, e.g., stainless steel.

At the Gloucester landfill, samples for compliance monitoring have been collected using dedicated, submersible pumps (QED Systems, Ann Arbor, Michigan) with PTFE (Teflon) bladders operated by compressed air or nitrogen which does not come into contact with the ground-water sample. The pumps, shown in Figure 12, are located at the depth of the well screen and can be isolated from the stagnant water in the well bore by inflating a packer system immediately above the pump. Generally, at least three well screen volumes are then pumped before sampling begins (Barcelona and Helfrich, 1986; Robin and Gillham, 1987).

Figure 13 shows the sequence of operations conducted in the field to collect samples and the subsequent distribution of aliquots for analytical purposes. Samples are collected in precombusted amber glass bottles, with no headspace for volatile organic samples, at a delivery rate of 100 mL/min or less. The bottles are allowed to overflow by at least 1.5 volumes then rapidly capped and stored at about 4°C until analysed (Barcelona et al., 1985). Sample bottles should not be rinsed out with the sample because of the potential for films of NAPLs to coat the bottles.

Preservation techniques for organic constituents of ground water differ from those for inorganics. In particular, it is inadvisable to subject samples collected for volatile organic analysis to vacuum filtration because of the potential volatilization of analytes. However, samples for analysis of volatile aromatic hydrocarbons, e.g., BTEX in Table 2, should be preserved with HCl acid (U.S. EPA, 1984a).

Analytical Organic Chemistry

The contaminants of concern at Gloucester were all organic, consequently their analysis at NWRI was by coupled gas chromatography-mass spectrometry (GC-MS). The principal components of GC-MS systems are shown in Figure 14. Essentially, the ground-water sample containing a mixture of chemicals was first concentrated (purge and trap) and then separated into individual chemicals in the capillary column of the GC, each compound yielding a peak on the chromatograph indicating a different retention time within the GC column. The chemicals elute from the GC and are then ionized into fragments of characteristic mass producing mass spectra that allow the identification of each chemical. It should be noted that GC alone cannot confirm the identity of a chemical because many chemicals have similar retention times in GC columns, consequently MS is an essential step in compound identification.

GC-MS analysis may be done in two modes - full scan or selected ion monitoring (SIM). In the first case, a range of masses (typically 45 to 450 atomic mass units) is acquired at the rate of one scan per second, i.e., every second a full mass spectrum is obtained and stored in the GC-MS's dedicated computer for later interpretation. In the second case, only those ions characteristic of the contaminants of concern are acquired, therefore SIM maximizes the sensitivity and selectivity of the GC-MS system. The analyses at Gloucester were done in the full scan mode which allowed the identification of several

unsuspected compounds (see below). Figure 15 shows the full scan or total ion current chromatogram for volatile organics in a sample of ground water from the Gloucester landfill.

Even when a full spectrum is acquired, quantitative analysis is done on extracted ions. One ion is selected for each compound for quantitation, i.e., each ion is a specific detector. The area under the curve of each peak of the chromatogram is integrated and compared to that of a standard. In addition, the area of one or two other characteristic ions is also integrated and their ratios compared to that of the primary ion; these are known as qualifying ions because they allow qualitative identification of the compound. Under the same operating conditions of the GC-MS, the ratios of qualifying to primary ions are constant. These, together with the retention times of the compounds in the GC column, comprise the criteria used to ensure the correct identification of the analytes.

The identification of the compounds contained within a sample is accomplished by comparing the spectrum of the unknown to spectra contained within a library stored within the GC-MS computer. There are two methods of comparison - forward and reverse searches. In a forward search, the spectrum of the unknown is compared to those contained in a library, e.g., that of the U.S. National Bureau of Standards (NBS) or a user-prepared library. The search yields a list of matches ranked in order of best fit. It is not uncommon to have matches of 90-95%, which would be considered confirmatory. However, when the matches are much less, the chemist must interpret the

differences and make tentative identifications. In a reverse search, a group of spectra contained in a user-created library is compared to all the spectra found in the sample. This type of search is commonly used in target compound or "priority-pollutant" analysis.

Forward searches will also identify non-priority pollutants that may be of considerable value in identifying the origin or pathway of the contaminated ground water (Swallow et al., 1988), however, they are more time consuming and therefore more costly than reverse searches. The latter have the advantage of being able to identify much lower quantities of a compound than is possible in a forward search.

Consequently, the advantages of the mass spectrometer as a detector over other detectors (ECD, FID, etc.) are that the quantitation is not affected by coeluting compounds and that the spectrum allows the unambiguous identification of the target compounds and tentative identification of non-target compounds.

QA/QC

The purpose of any quality assurance (QA) program is to reduce analytical measurement errors to agreed upon limits (quality control or QC) and to ensure that the analytical results have a high probability of being of an acceptable quality, a system known as quality assessment (American Chemical Society, 1983).

QA/QC programs need to be operated at two different levels. The first is within the laboratory to ensure that good lab practice is continually employed and that the results achieved are of consistent and acceptable quality. Secondly, external control by the client has to be implemented to ensure comparability of data obtained from different sources and over different time periods.

Two principal factors need to be assessed by the QA/QC program - precision and accuracy. They are defined as follows:

Precision: The degree to which data generated from replicate or repetitive measurements differ from one another (American Chemical Society, 1983).

Accuracy: The degree of agreement of a measured value with the true or expected value of the quantity of concern (Taylor, 1987).

Accuracy is more difficult to assess because, in many cases, certified reference materials, in particular for volatile organic compounds, are not available for many analytes and sample matrices (i.e., the totality of the sample, e.g., landfill leachate) and, furthermore, interlaboratory studies are infrequent. A suitable alternative would be to have another laboratory prepare a set of standards and spiked samples to be included with the rest of the samples. It should be stressed that split samples give little or no information about either precision or accuracy.

The principal QA/QC tools used by NWRI in its assessment of the Gloucester landfill were the employment of blanks, replicates,

surrogates and internal standards. As a rule, field or trip blanks were collected to evaluate external contamination of the sample, e.g., benzene from gasoline stored near the sample bottles, and the results corrected on the basis of such blanks. Replicates are important because they are the only measure one has of sample variability. Therefore, it is preferable to analyse one sample in triplicate and thus obtain statistics on the variability than to analyse two samples in duplicate, for which statistics cannot be estimated.

Surrogates, compounds that are analogs of the analytes and which are added to the sample prior to the extraction step, have replaced spiked samples as QA/QC tools. This is because the latter are impractical in that the laboratory does not usually know the expected concentration of the analyte a priori, therefore cannot spike the sample to a meaningful level. Most surrogates are deuterated (deuterium isotope labelled) analogs of the analyte of interest or chemical analogs, e.g., bromodichloromethane for volatile organics, assumed unlikely to be present in the sample. Since the surrogate is added prior to extraction, it is carried through the entire analytical procedure and its recovery reported as a measure of matrix effect and analytical error.

The final QA/QC tool used at NWRI was that of internal standards. These too may be deuterated analogs, however, they are added in the final step and are used for quantitation. Their purpose is to account for instrument variability from one sample to the next. Thus the peak area of the internal standard in the sample chromatogram

is compared to that in the standardization chromatogram and all concentrations are corrected for any variability. For volatile analysis, surrogates and internal standards are added together prior to the purging step.

RESULTS OF 1988 SAMPLING SURVEY

Sampling. ground water samples were collected during May 1988, from selected monitoring points within the monitoring well network. These points are shown in cross section in Figure 16. A set of five samples was collected from 5 cm i.d. piezometers (denoted by 124P...128P) each completed with stainless steel screens and dedicated Teflon bladder pumps and inflatable packers (QED Environmental Systems Inc., Ann Arbor, MI), which were inflated above the well screens to contact a stainless steel pipe extension of the screen. Prior to sampling, at least three well-screen volumes were discharged to waste. The remainder of the samples were obtained from bundle-type multilevel samplers (denoted by 16M, 37M etc., Cherry et al., 1983) using peristaltic pumps. Samples were collected in 25 or 40 mL amber glass vials and kept on ice until reaching the laboratory.

Measurements of pH and E_H were conducted in the field using combination glass and Pt electrodes fitted into air-tight flow cells (Jackson et al., 1985). Dissolved oxygen was determined using an Orbisphere Laboratories (York, ME) 2606 oxygen meter. Sulfide measurements were performed using a glass/sulfide electrode couple in

acidic solution (pH < 5) under closed, continous flow conditions (Gulens, 1985).

Analytical Methods. Volatile organic compound analysis was carried out following EPA method 624 (U.S. EPA, 1984b). A model 810 Unacon (Envirochem Inc., Kemblesville, PA) purge and trap was used to concentrate the analytes. This unit was interfaced to a Hewlett-Packard model 5970 GC-MSD. The analytical column was a 30 m DB-624 (J&W Scientific), 0.32 mm i.d., 1 µm film thickness, directly interfaced into the source of the MSD. The gas chromatograph was cooled to -5°C with liquid CO_2 to allow the analyses of the purgeable gases and programmed in two ramps to 140°C (-5° to 35°C at 10°/min and then at 4°/min). Difluorobenzene and chlorobenzene-d5 internal standards were added to samples of 10 mL immediately prior to purging. Mass acquisition was from 40 to 250 a.m.u. at 2 scans/sec, but quantitation was done on extracted ions specific for each analyte, using one qualifier ion for confirmation. Analytical standards were prepared from the purest available chemicals purchased from a variety of sources. Unknowns for which no standards were available were tentatively identified by comparison of their spectra to an NBS library containing 42,000 compounds.

Prior to 1983, the majority of the data was obtained using an HP7675 purge and trap interfaced to a GC-FID, with some of the analyses being confirmed on a Finnigan 4000 mass spectrometer. From 1984, a combination of FID and Hall detectors was used, until 1987, when the GC-MSD system described above was acquired.

Semi-volatiles and dioxins were analysed following the EPA 625 and 613 protocols (U.S. EPA, 1984c,d), except that a 10 L sample was extracted for dioxins. The analysis was done by Dr. R. Lao, RRETC, Ottawa, on a Finnigan 4000 GC-MS.

Semi-volatile Organic Compounds. Because most disposals of liquid organic chemicals at the site involved their incineration, and because pentachlorophenol wastes were thought to be included (i.e., as wood preservatives), the presence of chlorinated dibenzodioxins (CDDs) and dibenzofurans (CDFs) in ground water was suspected. The most prominent species detected in multilevel 37M-19 were octachlorodibenzodioxin and heptachlorodibenzofuran (Figure 17). No tetrachlorodibenzodioxins were detected. The total concentration for CDFs and CDDs were 43 and 59 pg/L respectively. The distribution pattern was similar to that observed for combustion of chlorinated organic products (Czucza and Hites, 1984).

Several GC-MS analyses were made in 1983 to detect semi-volatile contaminants directly beneath the SWC. Very few compounds were identified, most of which were naturally occurring fatty acids. The only priority pollutant identified was phenol at approximately 90 µg/L in 37P-1. An independent laboratory reported the pesticides silvex and 2,4-D at approximately 20 µg/L in 36W. More recently, analysis of multilevel sample 92M-15 using a dynamic thermal stripper (purge and trap at 95°C) did not reveal any anthropogenic semi-volatile compounds. This is not entirely surprising given the nature of the waste and also given the fact that most of these compounds have relatively high log K_{ow} values and would not be expected to migrate very far.

Volatiles Organic Compounds. Volatile organic compounds, most of them clearly identifiable laboratory solvents which survived incineration, constitute the bulk of the contaminants at Gloucester and are listed in Tables 8 and 9. Their behaviour will be discussed by groups, according to their mobility, with special attention paid to potential biotransformation products and their distribution. The possibility of the presence of a non-aqueous phase plume will also be discussed. Absolute concentrations (see Table 10) were generally much lower in May 1988 than during previous sampling periods (see Jackson et al., 1985), because of the large volume of contaminated ground water pumped from well 36W, used for the assessment of alternative treatment technologies, and returned to the aquifer through 36W (Figure 16). These operations only affected ground water quality directly beneath the SWC, where they caused a dilution of concentrations. Figures 18 and 19 show the migration patterns of selected volatile organic compounds.

a) Aliphatic Ethers. These compounds are of particular concern because of their rapid migration in the aquifer (Jackson et al., 1985; Patterson et al., 1985) as demonstrated by their presence in 126P and 127P (Table 10). The centre of mass of the diethyl ether and tetrahydrofuran plume moved from 34M to 21M from 1982 to 1988. The centre of mass for dioxane moved from 21M in 1982 to 47M in 1988. The movement of dioxane is of special concern because it has been recognized as a carcinogen. Although the U.S. EPA has not officially

classified this chemical, the drinking water concentration estimated to increase cancer risk by 10^{-6} is of 7 µg/L (U.S. EPA, 1988). The current analytical methodology (U.S. EPA, 1987) only achieves a practical detection limit of 150 µg/L, making the monitoring of the dioxane plume difficult. Also, this compound seems to have been refractory to biodegradation in the anoxic environment.

Analysis of samples collected from 85M, 68M and 134P during May 1989 indicates that a dioxane plume with maximum concentrations in excess of 1 mg/L has moved off site beneath the railroad tracks.

b) Aromatic Hydrocarbons. The concentration of these contaminants was very low, except for benzene. Therefore, it is most likely that there were no petroleum products other than laboratory solvents disposed of at the site, since one would then expect to find a larger number of alkyl benzenes. Benzene concentrations were reduced between 1982 and 1988 from 17 mg/L to 0.6 mg/L, the highest concentration still residing at the point of origin and low level residues (10 µg/L) having migrated as far as 54M (see Figures 18a and 19a).

This reduction is largely attributable to the experimental remedial operations and dispersion. From evidence gathered in a microcosm study (Wilson et al., 1986) and in an anoxic aquifer (Barker et al., 1987), biodegradation was likely to be slow with benzene as the most refractory compound.

c) Chlorinated Aliphatics. Plumes for these compounds as found in May 1988, are depicted in Figures 18b-e, 19b-e. The centres of mass for trichloromethane ($CHCl_3$), 1,1,1-trichloroethane (TCA) and

trichloroethene (TCE) occur at multilevel 67M, although that of the degradation product 1,1-dichloroethene (1,1-DCE) occurs at 37M-9.

Degradation of tetrachloroethene (PCE) and TCA appears to have occurred because of the presence of several metabolic intermediates such as 1,1-dichloroethane (1,1-DCA), chloroethene (VC), cis- and trans-1,2-dichloroethene (CIS and TRANS) and 1,1-DCE (Tables 8 and 10, Figure 20) in the same or in adjacent wells. Most samples contained both TCA and TCE and mixtures of their respective metabolites such that it is difficult to ascertain which pathway predominates, since some metabolites such as 1,1-DCE and VC are common to both. Possible inferences from spatial distribution are obscured because of the physical disturbances of the ground water. By integrating the concentration over the wells that have shown greatest contamination over the time period and considering the product distribution, it is possible to evaluate the results of biotransformation without having to consider differences in analytical methodologies, absolute concentrations, or the transport component. The product distribution for 1983 and 1988 are listed in Table 11.

From this data, it is possible to infer that degradation was indeed occurring since lesser chlorinated compounds increased relative to the more highly chlorinated compounds. Reductive dehalogenation reactions were prevalent, but elimination may have occurred to form 1,1-DCE from TCA, although this cannot be proven in the case of the chlorinated alkanes, because 1,1-DCE may have been formed from the reductive dehalogenation of TCE. Some of the products observed for

Freon degradation support the existence of this pathway (vide infra). The observed product ratio of CIS and TRANS was different than what has been reported previously (Wood et al., 1985; Parsons et al., 1984). In most samples, the two products were present in equivalent amounts, whereas these authors quoted a ratio of 25:1 for CIS:TRANS. CIS was found to be rapidly degraded in a microcosm study of methanogenic aquifer material (Wilson et al., 1986), hence it would not be expected to be present in higher concentration. One of the possible explanations for this difference is analytical, indeed some methods did not allow for good separation of the two isomers.

d) Freons. The presence of 1,1,2-trichloro-1,2,2-trifluoroethane (F113) had not been reported initially, probably because it is one of the first compounds to elute during gas chromatography. Consequently, its chromatography requires cryotrapping, which had not been used in the earlier analyses, i.e. prior to 1988. Its presence in a laboratory waste disposal site such as Gloucester is not surprising because this solvent is used in oil and grease determinations. Its mobility would be expected to be similar to that of PCE because their octanol/water partition coefficients are similar (log K_{ow} 2.57 for F113 vs. 2.60 for PCE; Roy and Griffin, 1985), and indeed it had migrated to the same area. It seems much more refractory to biological transformation and is now the compound found in largest concentration at the site (2.7 mg/L in 67M-11), more than five times the TCE and PCE concentrations. Unfortunately, no historical disposal

data is available for F113, which means that its attenuation over time cannot be assessed.

1,2-dichloro-1,2,2-trifluoroethane (F123a) was also found in the same samples, but at much lower concentrations (10-50 µg/L) than F113. F123a could conceivably have been present as an impurity in F113. According to the manufacturer (S. Lauridsen, Dupont Canada, Maitland, Ontario, pers. comm.), it can comprise up to 0.07% of the product in the synthesis reactor, prior to initial purification. The concentration of F123a found in the samples comprised from 0.5 to 50% of the F113 concentration, far in excess of the possible impurity level. Furthermore, a sample of the technical products as sold to redistillers was analysed and found not to contain any detectable F123a.

F123a could also have been formed by pyrolysis. Indeed F123a has been identified in the combustion products of F113 in the presence of steel, copper and lubricating oil (Srinivasan et al., 1984). The authors did not confirm it as a product of pyrolysis, referring to the fact that it could be found as an impurity in the technical product. The major product of combustion they observed was chlorotrifluoroethene (F1113), which was also identified in multilevel 67M-11 at an estimated concentration of <10 µg/L. Therefore, the observed product ratio (F123a:F1113) was the reverse of what would be expected from combustion of F113 alone.

F123a could also have been the product of in-situ microbial degradation. In support of this hypothesis was the fact that in spite

of their predicted higher mobility, F123a and F1113 were found in the same multilevels as F113, and their ratio was not consistent with the known pyrolysis products. Furthermore, these compounds were found in the midst of an anoxic zone where there was evidence of sulfate reduction. The formation of F1113 from F113 might follow the same metabolic pathway as the formation of VC from TCA (Figure 20a). The first reaction is one of reductive dehalogenation, followed by the elimination of HCl to form the olefin. The mechanism of the reaction has been postulated as involving free-radical formation involving cytochrome P-450 in the case of the metabolism of 2-bromo-2-chloro-1,1,1-trifluoroethane (Monig and Asmus, 1984). We also tentatively identified 1,1,2-trichloro-1,2-difluoroethane (F122a) which, if it was formed from F113, would be most likely to arise via a free-radical mechanism which would tend to be less specific than a substitution mechanism, in which the loss of F over Cl is not favored.

F113 has been gradually replacing PCE as a dry-cleaning solvent and is widely used in the electronic industry, where it has caused substantial ground water contamination (Esau and Chesterman, 1988). This work shows that it may be even more of a threat to ground water, because it appears more persistent than PCE. It was found to act as a co-carcinogen with polyaromatic hydrocarbons without any inference to the possible pathway (Mahurin and Bernstein, 1988). Both F113 and F123a were found to be non-mutagenic in Ames tests, however, the authors caution this may not be a valid indicator of mammalian toxicity. This is because these compounds could be analogous to

1-fluoro-2,2,2-trifluoroethane which, although it was found to be nonmutagenic in Ames test, was found to be carcinogenic in an animal study (Longstaff et al., 1984). F1113 is used in its gaseous form in the polymer industry and is classified as highly toxic (268 mg/kg oral; Sax, 1984); its presence in ground water is, therefore, of concern.

e) Sulfur containing compounds. As noted above, that part of the outwash aquifer associated with the chloroalkane/ene plume is a sulfide redox zone and the presence of organo-sulfur compounds was suspected from their characteristic odour. The organo-sulfur compounds identified (Table 9) were: dimethyl sulfide, diethyl sulfide, methyl ethyl sulfide and tetrahydrothiophene in monitors 37M, 67M and 125P.

Because these were found in areas where chlorinated organic solvents were present in high concentrations, the possibility of reaction between the two species was considered, but no chlorinated organo-sulfur products were identified. Indeed, although sulfides may react with halogenated aliphatics to form mercaptans (Schwarzenbach et al., 1985), and, although sulfur nucleophiles are more reactive than their oxygen analogs, their relative concentrations in aqueous systems are so much smaller that these reactions are seldom observed (Vogel et al., 1987). The formation of chloroalkylsulfones, which are resistant to biodegradation, has been reported (Headley et al., 1987), but their presence has been attributed to discharge from biologically treated kraft mill effluents, which means that their occurrence would constitute an exception rather than the norm in ground water systems.

DISCUSSION OF CONTAMINANT TRANSPORT MECHANISMS

The widespread migration of chloride and organic chemicals shown in Figures 9, 10, 11, 18, and 19 is due to solute transport by ground water. It has been shown elsewhere (Jackson et al., 1985) that this solute transport was affected by sorption of the organic chemicals by aquifer materials (f_{oc} = 0.06%). Due to hydrophobic processes, sorption causes the organic solutes to migrate at differing mean velocities, which are functions of the octanol-water partition coefficient (K_{ow}) of the compound. Solutes with K_{ow} values that are relatively small, such as the aliphatic ethers (i.e., 1,4-dioxane, diethyl ether and tetrahydrofuran), have travelled the farthest. For example, large concentrations of oxygenated solvents are present at 126P and 127P, but aromatic hydrocarbons and chlorinated aliphatics have been retarded in their transport and are to be found mainly beneath the SWC (Figures 18 and 19). Solute transport and sorption produce retardation factors (ratio of ground water velocity to contaminant velocity) that vary from 1.6 (1,4-dioxane) to >20 (carbon tetrachloride).

In addition to aqueous-phase solute transport, there is the possibility that organic contaminants may migrate through the unsaturated and saturated zones of the ground water flow system as separate phases whose behaviour is a function of their relative saturation, density and viscosity. Of particular concern are those

organics that constitute such as the halogenated aliphatics. These may be present in the flow system as either pools of liquid chemical ("free product") in contact with some relatively impermeable hydrostratigraphic unit (e.g., bedrock or silt) or as ganglia (residual droplets) left behind in pores as the DNAPLS sank through the aquifer system (Schwille, 1984; Sitar et al., 1987; Schwille, 1988; Feenstra and Cherry, 1988).

The occurrence of DNAPLs in aquifer systems is particularly hard to verify unless the fouling of downhole equipment or the saturation of cores is observed. However, these instances would be the exceptions rather than the rule, due to the relatively small size of most pools or ganglia and the relatively large distances between boreholes. Typically, DNAPLs produce concentrations in nearby monitoring wells that are less than 10% of their aqueous solubility (Feenstra and Cherry, 1988). Concentrations approaching the solubility limit are not measured in ground waters due to the heterogeneous spatial distribution of ganglia or pools in the subsurface, mass transfer constraints on their dissolution, and the dilution of the dissolved components during ground water transport, transformation and sampling (Feenstra and Cherry, 1988; Anderson et al., 1987; Hunt et al., 1988a).

The likelihood of pools of DNAPL being present in the outwash aquifer immediately beneath the SWC would appear remote. Figures 21 and 22 show vertical profiles for three DNAPLs and benzene at multilevel samplers within and close to the SWC in 1982, 1984 and 1988. The chlorinated aliphatics show no persistent nor monotonic increase

with depth as might be expected if a DNAPL pool had sunk vertically to the bottom of the aquifer. The relatively large concentrations of volatile organic chemicals measured in 1982 and 1984 are not observable in the 1988 profile, due to diluting effects of the experimental pump and treat operations and of natural ground water dispersion and biotransformation. Furthermore, a bedrock monitoring well (37P-3, see Figure 16), located beneath the SWC and hydraulically connected to the outwash aquifer, remains uncontaminated. Had a DNAPL pool been present, the termination of experimental pumping in 1986 would have been followed by an increase in the concentration of contaminants dissolved in ground water.

However, this conclusion may not apply to those parts of the outwash aquifer adjacent to the SWC that are not confined by the aquitard (Unit D). The samples from the 1988 survey of the SWC, which were analysed by GC-MSD with cryotrapping, identified the presence of large quantities of the freon, F113, and its possible transformation products. At multilevel sampling point 67M-11, some 20 m south of the SWC, the concentration of F113 was approximately 2% of its aqueous solubility at 10°C, a percentage indicative of the possible presence of DNAPLs. Furthermore, concentrations of other organic residues have exceeded 1% of their solubility as is shown by chloroform and PCE in Figure 23.

This analysis suggests that, following disposals of F113, the freon may have migrated as viscous fingers (Schwille, 1988; Anderson et al., 1987) through the water-table aquifer (Unit E) and,

by some pathway, through or around the aquitard (Unit D) and into the outwash aquifer (Unit C). A thin, clayey gravel layer was identified in drilling logs at an elevation immediately below sampling point 67M-11. This indicates that a small pool or extended ganglia may well exist at this location and be sitting on a low permeability strata quite common to this type of glaciofluvial sediment. Furthermore, this monitor also contains the maximum concentrations of other volatile organic chemicals (1,2-dichloroethane, chlorobenzene, TCA, TCE and PCE) observed at the site; these may have been part of the same disposal. This scenario implies that ganglia or pools exist along the pathways taken by the viscous fingers, particularly at the contact between the water-table aquifer and the aquitard.

CHAPTER 4

MOBILITY OF ORGANIC PRIORITY POLLUTANTS IN THE OUTWASH AQUIFER

The migration of the organic priority pollutants discussed in Chapters 2 and 3 are mainly the result of solute transport due to ground-water flow through the outwash aquifer as well as solute sorption, resulting in "chromatographic dispersion" (Patterson et al., 1985). In order to predict the rates of migration of the organic solutes, two types of experiments have been conducted to determine the velocity of the organic solutes relative to that of a conservative or non-reactive tracer, e.g., chloride or tritium.

The first of these experiments was a field test within the outwash aquifer, known as the "Purge Well Test", conducted to test the efficacy of using a pump and treat system to decontaminate the outwash aquifer. The second type of experiment involves the use of laboratory columns of Gloucester aquifer sands to estimate the migration rates. The parameter of interest in both experiments is known as the retardation factor, R_f, which is defined by Freeze and Cherry (1979, p. 404) as:

$$R_f = 1 + (\rho_b K_D/n)$$

where ρ_b is the bulk density, K_D is the solute distribution (or partition) coefficient and n is the porosity of the aquifer sediments. The distribution coefficient is merely the slope of a linear

adsorption isotherm for the organic solute and aquifer sediment under consideration. In the case of a non-reactive solute, $K_D = 0$ and $R_f = 1$.

Thus the retardation factor allows one to estimate the mobility of a particular organic solute in a particular aquifer material from the following expression:

$$V_C = V_{GW}/R_f$$

where V_C and V_{GW} are the mean velocities of the contaminant solute and the ground water, respectively. This information is crucial in predicting the furthest extent of contaminant migration and in assessing the duration of aquifer remediation.

PURGE WELL TEST

In the fall of 1983 a test was undertaken by Whiffin and Bahr (1985) to study the decontamination of the outwash aquifer using purge wells. The site chosen for this experiment was in the middle of the chloride plume, i.e., near multilevel 54, see Figure 6. The test was conducted by injecting uncontaminated, tracer-labelled ground water into a fully-penetrating well at the rate of about 140 L/min for the first day, about 95 L/min for the following 4 days and 140 L/min for the sixth and final day, while withdrawing at the same rates from a second, similar well 5 m away (see the inset in Figure 24). In

between the two wells was situated a multilevel sampler, M1, from which samples were taken throughout the 6 day period of the test. These were analysed for two injected tracers (iodide and a fluorinated benzoic acid), both presumed to be nonreactive, and for three oxygenated solvents present in that part of the aquifer -- 1,4-dioxane, tetrahydrofuran and diethyl ether.

Retardation factors (R_f) for each of the contaminants were estimated by dividing the $C/C_0 = 0.5$ time for the disappearance of the chemical due to its withdrawal or "purging" by the $C/C_0 = 0.5$ time for the arrival of the nonreactive tracers at the multilevel. These R_f values are shown in Table 12 along with values determined independently from (1) plume mapping (Patterson et al., 1985) and (2) the equation of Schwarzenbach and Westall (1981) and the retardation equation. The close agreement between the R_{f1} and R_{f2} values suggest that the plume-length retardation factors adequately describe the relative mobility of the organic contaminants.

The differences between the R_f value obtained by the equation of Schwarzenbach and Westall (i.e., R_{f4}) with the other values indicate that this predictive equation will yield results that are increasingly in error as compounds with larger K_{ow} values are considered. This is likely due to sorption of the organic solutes on mineral surfaces, a factor not incorporated into the prediction equation (Curtis et al., 1986).

The behaviour of the organic contaminants during desorption is shown in Figure 24, in which the time axis is replaced by the number of pore volumes pumped through the test section of the aquifer (a pore

volume is defined by Whiffin and Bahr (1985) as "the volume of water required per unit aquifer thickness to displace the interstitial ground waters located between the injection and monitoring points as defined by the breakthrough of the 50% concentration level for the nonreactive tracer"). While dioxane seems to be purged rapidly from the test section within the aquifer, tetrahydrofuran (THF) and diethyl ether (DEE) require longer purge times and display pronounced tailing which is probably due to nonequilibrium sorption/desorption reactions.

This tailing is consistent with that expected in the case that the transport of these contaminants is influenced by a linear reversible sorption/desorption process that does not proceed at a rate sufficient to reach local equilibrium during desorption at these flow velocities (3 m/day at M1 or roughly 50 times the natural ground-water flow velocity). THF also exhibits an abrupt increase in concentration following the decrease in flow rate after 24 hours. The rate limiting process may be either the sorption mechanism itself or a physical diffusion process (Bahr, 1989).

LABORATORY COLUMN TESTS

The laboratory soil column offers the most practical and inexpensive method of estimating R_f in sand and gravel aquifers, e.g., Reynolds et al., 1982; Lee et al., 1988a. It requires that the laboratory has access to a variety of variable flow rate pumps, an effluent fraction collecting system and some means for analyzing the

effluent, e.g., GC or liquid scintillation counter. In estimating the retardation factor of a volatile organic chemical (VOC), such as diethyl ether or tetrachloroethene, the VOC is treated as a tracer and injected into ground water flowing into the column.

The method used to estimate R_f values for VOCs in the Gloucester aquifer involves packing a short glass column (10-30 cm long) with aquifer material to the bulk density of the aquifer. The column, shown in Figure 25, is fitted with end plates to allow the entry and elution of the tracer and ground water. It is then filled slowly from below with deaerated ground water so that air bubbles are not trapped inside it.

Figure 26 shows the complete soil column apparatus, which is maintained at 10°C in a cold room. The ground water supply is taken from multilevel 84M-6, upgradient of the SWC, and is passed through a 0.2 µm filter. It is then pushed into the column via a tubing pump, which regulates the velocity of the ground water. Prior to its entry into the column, the tracers (i.e. organic contaminants and iodide) are added via a syringe pump and this solution passes through a mixer containing stainless steel balls. The input concentration, C_0, is measured by taking a sample in a Tedlar bag from a port immediately before the column. The effluent from the column is discharged into evacuated Tedlar bags each fitted with a Teflon on/off valve. The filling of the bags is controlled by a valve sequencer. The bags are disconnected after filling and a syringe is used to remove a sample of the effluent for injection into a purge and trap GC/MSD. The plot of

the normalized effluent concentration (C/C_0) versus time is known as the breakthrough curve or BTC.

The retardation factor can be estimated by several methods. The simplest of these is to compare the arrival times of the organic tracer and that of a non-reactive tracer, such as iodide, using the values at $C/C_0 = 0.5$ (or $C/C_{max} = 0.5$ if C_0 is not reached). However, this method ignores the physical and chemical nonequilibrium processes that cause substantial tailing and assymetry in the BTC (e.g., Lee et al., 1988a). As Bahr (1989) has shown, these processes are important in pump and treat operations and should be incorporated in the measured value of R_f.

Parker and van Genuchten (1984) have developed a suite of solute transport models to fit to BTCs using optimization methods to determine R_f and other parameters. Two of the models are of interest in the present circumstances. The first fits a linear equilibrium assumption (LEA) model to the BTC and thus assumes that sorption of the organic tracer is instantaneous, reversible and follows a linear adsorption isotherm. The second, a nonequilibrium model, treats the sorption reaction as involving two different types of sites or regions in the aquifer material: an LEA fraction and a second fraction for which sorption is considered to be time dependent or rate limited.

A number of tests were run using the apparatus shown in Figure 26 with a sample of the Gloucester outwash aquifer taken from 15 m depth, 100 m east of the SWC. The column was packed to a bulk density of 1.76 g/cm^3 with a porosity of 0.32 and a pore volume of 205 cm^3.

Analysis of 20 samples of the aquifer material indicated a mean organic carbon content of 0.07% and a standard deviation of 0.03%. This value is considerably lower than those reported for the Gloucester outwash aquifer materials by Jackson et al. (1985); these earlier values are now thought to be incorrect.

Results from one of the column tests (#3) are shown in Figures 27 and 28 and summarized in Table 12. Ground water flowed through the column with an average linear velocity of 45 cm/day. The length of the pulse of the organic tracers was 0.65 pore volumes. Estimates of R_f taken from the pore volume value when $C/C_0 = 0.5$ are low compared with the fitted results from the nonequilibrium model (CXTFIT, model 4) of Parker and van Genuchten (1984). Because of the failure of this model to reproduce the examples in Parker and van Genuchten (1984), these latter values are considered suspect. A code to compute the centre of mass (CofM) of the BTC gave estimates midway between those two values. These CofM values are considered to be the most reliable on the basis of their similarity to the estimates of Patterson et al. (1985) for trichloromethane (TCM or chloroform) and 1,1-dichloroethene (11DCE), compounds not used by Patterson et al. to derive their prediction equation.

It is noteworthy that the method of Schwarzenbach and Westall (1981) gives poor estimates when compared to the field R_f values. This may be expected because the method was developed from correlations with compounds having log K_{ow} values from 2.6 to 4.7 and for aquifer materials with at least 0.1% organic carbon. Furthermore, it

is evident from Figure 28 that the compounds elute in approximately the order of their increasing log K_{ow} value. However, coelution of 1,2-dichloroethane (12DCA) and trichloromethane occurs despite their log K_{ow} values being one-half log unit different. Finally, the LEA model (#2) of Parker and van Genuchten (1984) provides better estimates of R_f for the only slightly retarded aliphatic ethers with their symmetric BTCs (see Figure 27) than the nonequilibrium model (#4) does for the strongly retarded VOCs, which display asymmetric BTCs as shown in Figure 28.

The data summarized in Table 12 suggests that laboratory column tests give reasonable estimates of the retardation factor for a organic solute, when compared to those estimated in the field.

CHAPTER 5

DISCUSSION OF REMEDIAL OPTIONS

The alternatives for remedial action commonly used in the U.S.A. and Canada are listed in Table 13. These were considered by Graham Engineering Consultants and Water and Earth Sciences Ltd. (GEC, 1985b) in their assessment of remedial measures at the Gloucester landfill.

Of the management options, GEC considered the "no action" alternative to be unacceptable on the basis of the results of an endangerment assessment (Jackson et al., 1985) which showed that the upper limit on lifetime cancer risk from drinking contaminated ground water from the site was much less than 1 in 1000. GEC did recommend that alternative water supplies be developed for the community immediately east of the CPR tracks shown in Figure 6. In 1985, a pipeline was laid to supply these homes and industries with water from the City of Ottawa system.

A second management option recommended and implemented during 1988/89 was source removal. This involved the excavation of eight one-ton lugger boxes of bulk wastes consisting primarily of pesticide-contaminated soils and fifty-two 58-gallon (218 L) drums of liquid and solid chemical and biochemical wastes (MacLaren Engineers, 1988).

Containment alternatives considered included the installation of impermeable slurry walls or grout curtains around the zone of highest contamination. These techniques were rejected principally because the

limestone bedrock underlying the plume is very permeable and is hydraulically connected to the overlying outwash aquifer (Jackson et al., 1985). In similar conditions at the Sylvester site in Nashua, N.H., the remedial slurry wall and cap system was "much less effective than anticipated" (OTA, 1985) due to leakage through the bedrock. Consequently complete isolation could not be assumed at the Gloucester landfill.

Aquifer decontamination was selected as the most appropriate alternative involving "pump and treat" scheme which, it is estimated, will need to operate five years, and involve four purge wells situated along the axis of the plume (GEC, 1985b). Ground water discharge will be treated by a bubbler system to remove dissolved iron, air stripping and granular activated carbon adsorption. Treated water will be returned to the aquifer by five recharge wells. Tentative locations for the purge and recharge wells are shown in Figure 29. The four purge wells are planned to operate at a total rate of 120 gpm (655 m^3/day) based upon aquifer tests results which should ensure hydraulic containment of the plume (GEC, 1985b). The cost of this operation is estimated to be of the order of C$6 million (US$5 million) in current dollars.

Prior to the initiation of a pump and treat operation, the presence of immiscible organic phases in the aquifer must be assessed. Such dense, non-aqueous phase liquids (DNAPLs) released by waste disposal operations or by spills tend to migrate to depth leaving residual droplets of liquid within the unsaturated and

saturated zones. If DNAPLs are present in sufficient volume, pools may form on relatively impermeable strata within the aquifer or at its base (see Schwille, 1984, 1988; Feenstra and Cherry, 1988). Residual droplets ("ganglia") and pools are a long-term source of aquifer contamination. If they remain in an aquifer, decontamination may not be achieved by pump and treat operations over the short term (Hunt et al., 1988a) or recontamination of a supposedly decontaminated aquifer may occur once the pump and treat operation is suspended.

As discussed in Chapter 3, the presence of DNAPLs beneath the SWC at Gloucester is uncertain. Concentrations of tetrachloroethene (PCE) in ground waters immediately beneath the SWC exceeded one percent of its aqueous solubility during 1982/83, as shown in Figure 23; this value is a rough indicator of the presence of DNAPLs in an aquifer (Hunt et al., 1988a). However, since that time, concentration of most volatile organic contaminants have declined by a factor of about 100 (see Table 14 and Figures 30-33) suggesting the absence of DNAPL pools. Furthermore, Figures 21 and 22 show no increase in contamination with depth as would be expected should a DNAPL pool have sunk through the aquifer. In addition, monitoring well 37P-3 in the underlying bedrock remains free of contamination despite being hydraulically connected to the outwash aquifer. Therefore, it seems entirely possible that DNAPLs in limited quantities did enter the outwash aquifer in 1982/83 or before, and still exist as emulsified droplets or ganglia but not as pools.

Experimental pump and treat operations were conducted during 1984 and 1988 in the SWC with withdrawal and recharge wells completed in the outwash aquifer beneath the SWC. Results suggest that the residual contaminants can be removed from the aquifer, although not necessarily from the overlying aquitard which may require excavation. Figures 30-33 show the substantial reduction in benzene, chloroform, 1,1,1-trichloroethane and trichloroethene between 1984 and 1988.

In order to demonstrate the effectiveness of decontaminating the outwash aquifer by purge wells (i.e., the "pump and treat" option), the Purge Well test was conducted (Whiffin and Bahr, 1985; Bahr, 1989) at a site (54M) near the centre of the contaminant plume. As discussed in Chapter 4, this test involved the introduction of uncontaminated ground water into the contaminated aquifer via an injection well and the withdrawal of a similar volume of contaminated ground water from another well 5 m away. Operation of this system resulted in purging of the contaminated aquifer between the two wells (see Figure 24). A second test of this type was conducted in a shallow plume of organic contaminants in the unconfined aquifer (Unit E) with similar results (WESA, 1988).

Analysis of the disappearance curves in Figure 24 indicates that there is a log-linear relationship (Figure 34) between the number of pore volumes withdrawn from the aquifer to obtain 90% apparent decontamination of each of the organics and their respective K_{ow} values. This site specific relationship can be extrapolated (i.e., beyond ten pore volumes) to obtain an estimate of the number of pore volumes

required to decontaminate parts of the aquifer containing contaminants not found in the test area. It must be noted that the values in Figure 34 are minimum estimates of the number of pore volumes required to decontaminate this particular aquifer under the hydraulic conditions of the test. Because local chemical equilibrium was not attained everywhere during the test, large volumes of relatively dilute ground water were pumped from the aquifer. Bahr (1989) has concluded "that pumping may be more efficient, in terms of volumes of clean water required for injection and volumes of contaminated water removed for treatment, if lower flow rates are used". By so doing, more time is available to allow the relatively slow desorption of the organic chemicals.

Because of the high cost of pump and treat operations, there is much interest in developing lower cost alternatives. The most promising of these is in-situ biological remediation. This technique, generally known as biorestoration, involves the prior removal of DNAPL pools and the subsequent injection of oxygen and nutrients into a plume of dissolved aromatic hydrocarbons to stimulate the native microbial populations to metabolize or cometabolize the organic contaminants (Wilson et al., 1986; Lee et al., 1988b). Because in-situ biorestoration of halogenated hydrocarbons is still under development (Semprini et al., 1988), it was not considered as a remedial option for the Gloucester aquifer. However, it is anticipated, that towards the end of the five years of purging the contaminated aquifer, in-situ

biorestoration may become available to clean up the residual contamination that will undoubtedly remain. The biological techniques are potentially cost effective methods for reducing residual contamination to compliance levels, which, in this case, are similar to EPA's maximum contaminant levels.

Other in-situ techniques, such as steam injection (Hunt et al., 1988a, b), the use of surfactants (Vigon and Rubin, 1989) and enhanced solubilization (Zenon, 1986), all show promise but as yet are unproven.

CONCLUSIONS

An exhaustive assessment has been conducted of the organically-polluted outwash aquifer beneath the Gloucester Landfill and potentially applicable alternative strategies for remediation have been evaluated. It is concluded that: (Jackson et al., 1989; Jackson and Patterson, 1989)

1. the installation of a slurry wall system to contain the organic pollutants is unsuitable because of the very permeable nature of the bedrock;
2. pools of DNAPL chemicals are unlikely to be present within the outwash aquifer although emulsified DNAPL chemicals probably still exist as ganglia;
3. the aquitard immediately beneath the SWC may contain substantial quantities of DNAPLs and should be investigated with regard to its possible excavation;

4. a "pump and treat" operation should be successful in removing most of the dissolved and sorbed organic chemicals in the outwash aquifer provided that the purge and recharge wells are placed in the optimal locations; and
5. in-situ biorestoration may be required to complete the decontamination of the aquifer following the "pump and treat" operations.

REFERENCES

American Chemical Society Committee Report. 1983. Principles of Environmental Analysis, Analytical Chemistry, 55(14):2210-2218.

Ames, B.N., Magaw, R. and Gold, L.S. 1987. Ranking possible carcinogenic hazards. Science, 236:271-280.

Anderson, M.R., Johnson, R.L. and Pankow, J.F. 1987. The dissolution of residual dense non-aqueous phase liquid (DNAPL) from a saturated porous medium. In: Proc. NWWA/API Conf. on Petroleum Hydrocarbons and Organic Chemicals in Ground Water, Houston, TX, published by NWWA, Dublin, OH, pp. 409-427.

Bahr, J.M. 1989. Analysis of nonequilibrium desorption of volatile organics during field test of aquifer decontamination. Journal of Contaminant Hydrology, 4:205-222.

Banerjee, S. 1984. Solubility of organic mixtures in water. Environ. Sci. Technol., 18(8):587-591.

Barcelona, M.J., Gibb, J.P., Helfrich, J.A., and Garske, E.E. 1985. Practical guide to ground-water sampling. Illinois State Water Survey, Champaign, Illinois.

Barcelona, M.J. and Helfrich, J.A. 1986. Well construction and purging effects on ground-water samples. Environ. Sci. Technol., 20(1):1179-1184.

Barker, J.F., Patrick, G.C. and Major, D. 1987. Natural attenuation of aromatic hydrocarbons in a shallow sand aquifer. Ground Water Monitoring Review, 7(1):64-71.

Barrio-Lage, G., Parsons, F.Z., Nassar, R.S., and Lorenzo, P.A. 1986. Sequential dehalogenation of chlorinated ethenes. Environ. Sci. Technol., 20(1):96-99.

Champ, D.R., Gulens, J. and Jackson, R.E. 1979. Oxidation-reduction sequences in ground water flow systems. Canadian Journal of Earth Sciences 16(1):12-23.

Cherry, J.A., Gillham, R.W., Anderson, E.G., and Johnson, P.E. 1983. Migration of contaminants in groundwater at a landfill: A case study. 2. Groundwater Monitoring Devices. J. Hydrol. 63:31-49.

Curtis, G.P., Roberts, P.V. and Reinhard, M. 1986. A natural gradient experiment on solute transport in a sand aquifer. 4. Sorption of organic solutes and its influence on mobility. Water Resources Research, 22(13):2059-2067.

Czucza, J.M. and Hites, R.A. 1984. Environmental fate of combustion generated polychlorinated dioxins and furans. Environ. Sci. Technol., 18(6):444-450.

Esau, R.R. and Chesterman, D.J. 1988. Ground water contaminatoin issues in Santa Clara County, California: A perspective. In "Hazardous Waste Site Management: Water Quality Issue", National Academy Press, Washington, D.C., pp. 120-136.

Feenstra, S. and Cherry, J.A. 1988. Subsurface contamination by dense non-aqueous phase liquid (DNAPL) chemicals. In: Proceedings of the International Association of Hydrogeologists, Halifax, Nova Scotia. 15 pp.

Freeze, R.A. and Cherry, J.A. 1989. What has gone wrong. Ground Water, 27(4):458-464.

Freeze, R.A. and Cherry, J.A. 1979. Ground water. Prentice Hall Inc., New Jersey.

GAO. 1988. Ground water conditions at many land disposal facility remain uncertain. U.S. General Accounting Office, Washington, D.C., 53 p.

GEC (Graham Engineering Consultants). 1985a. Waste Types and Quantities Disposed in the Gloucester Landfill Site and Special Waste Compound. Ottawa, Ontario, 18 p.

GEC. 1985b. Gloucester Landfill Waste Site, Problem Definition and Remedial Alternatives. A.J. Graham Engineering Consultants Ltd., Gloucester, Ontario, 177 p.

Gold, L.S. and 11 others. 1984. A carcinogenic potency database of the standardized results of animal bioassays. Environ. Health Perspective, 58:9-319.

Grisak, G.E., Jackson, R.E. and Pickens, J.F. 1978. Monitoring ground water quality: The technical difficulties. In: Establishment of Water Quality Monitoring Programs, L.G. Everett and K.D. Schmidt, (Eds.), AWRA, Minneapolis, Minnesota, pp. 210-232.

Gulens, J. 1985. A portable monitor for measurement of dissolved sulfide based on the glass/sulfide-selective electrode couple. Water Res., 19(2):201-204.

Headley, J.V. 1987. GC/MS identification of organosulphur compounds in environmental samples. Biomed. and Environ. Mass Spec., 14:275-280.

Hirschhorn, J.S. and Oldenburg, K.U. 1988. Are we cleaning up? An assessment of Superfund. Chemical Engineering Progress, 84(12):55-65.

Horvath, A.L. 1982. Halogenated hydrocarbons: solubility, miscibility with water. Marcel Dekker, New York, pp. 473-573.

Hunt, J.M. 1979. Petroleum Geochemistry and Geology. W.H. Freeman and Company, San Francisco.

Hunt, J.R., Sitar, N. and Udell, K.S. 1988a. Nonaqueous phase liquid transport and cleanup. 1. Analysis of mechanisms. Water Resources Research, 24(8):1247-1258.

Hunt, J.R., Sitar, N. and Udell, K.S. 1988b. Nonaqueous phase liquid transport and cleanup. 2. Experimental studies. Water Resources Research, 24(8):1259-1269.

Jackson, R.E. and Patterson, R.J. 1982. Interpretation of pH and Eh trends in a fluvial-sand aquifer system. Water Resour. Res. 18(4):1255-1268.

Jackson, R.E., Patterson, R.J., Graham, B.W., Bahr, J., Belanger, D., Lockwood, J., and Priddle, M.W. 1985. Contaminant Hydrogeology of Toxic Organic Chemicals at a Disposal Site. Gloucester, Ontario. 1. Chemical Concepts and Site Assessment. IWD Scientific Series No. 141, Ottawa, Ontario, 114 p.

Jackson, R.E., Crowe, A.S., Lesage, S., and Priddle, M.W. 1989. Aquifer contamination and restoration at the Gloucester Landfill, Ontario, Canada. In: Ground water Contamination, Proc. Symp., Third IAHS Scientific Assembly, Baltimore, MD, May, 1989, IAHS Publication No. 185, pp. 181-188.

Jackson, R.E. and Patterson, R.J. 1989. A remedial investigation of an organically polluted outwash aquifer. Ground Water Monitoring Review, 9(3):119-126.

Keith, L.H. and Telliard, W.A. 1979. Priority Pollutants. 1 - A Perspective View. Environ. Sci. Technol., 13(4):416-423.

Lee, L.S., Rao, P.S.C., Brusseau, M.L., and Ogwada, R.A. 1988a. Nonequilibrium sorption of organic contaminants during flow through columns of aquifer materials. Environ. Toxicol. Chem., 7:779-793.

Lee, M.D., Thomas, J.M., Borden, R.C., Bedient, P.B., Wilson, J.T., and Ward, C.H. 1988b. Biorestoration of aquifers contaminated with organic compounds. CRC Critical Reviews in Environmental Control, 18(1):29-89.

Lesage, S. and Jackson, R.E. 1989. Degradation pathways of halogenated ethanes and ethenes in ground water. Presented at the 19th Annual Symposium on the Chemistry of Contaminants in the Environment. Jekyll Island, Georgia, May 21-24.

Longstaff, E., Robinson, M., Bradbrook, C., Styles, J.A., and Purchase, I.F.H. 1984. Genotoxicity and carcinogenicity of fluorocarbons. Toxicology and Applied Pharmacology, 72:15-31.

MacLaren Engineers. 1988. Gloucester Landfill Waste Site cleanup - Phase I. Stage I - Field investigation summary report. Willowdale, Ontario, Canada.

Mackay, D. and Cherry, J.A. 1989. Groundwater contamination: pump and treat remediation. Environ. Sci. Technol., 23(6):630-636.

Mahurin, R.G. and Bernstein, R.L. 1988. Fluorocarbon-enhanced mutagenesis of polyaromatic hydrocarbons. Environmental Research, 45:101-107.

Martel, F. 1988. Groundwater contamination by organic compounds in Ville Mercier: new developments. In: Proceedings of the Second International Conference, NATO/CCMS Demonstration of Remedial Action Technologies for contaminated land and groundwater, Bilthoven, The Netherlands, 7-11 November 1988, pp. 144-177.

Michel, F.A., Kubasiewicz, M., Patterson, R.J., and Brown, R.M. 1984. Ground water flow velocity derived from tritium measurements at the Gloucester Landfill site, Gloucester, Ontario. Water Poll. Res. J. Canada, 19(2):13-22.

Monig, J. and Asmus, K.D. 1984. One-electron reduction of halothane by free radicals. J. Chem. Soc. Perkin Trans. II, 12:2057-2063.

OTA (Office of Technology Assessment). 1984. Protecting the nation's ground water from contamination. OTA, Washington, D.C.

OTA (Office of Technology Assessment). 1985. Superfund Strategy, OTA-ITE-253, Washington, D.C.

Parker, J.C. and van Genuchten, M.Th. 1984. Determining transport parameters from laboratory and field tracer experiments. Virginia Agricultural Experiment Station, Bulletin 84-3, Blacksburg, VA.

Parsons, F., Wood, P.R. and Demarco, J. 1984. Transformations of tetrachloroethene and trichloroethene in microcosms and groundwater. J. Amer. Water Works Assoc., 76:56-59.

Patterson, R.J., Jackson, R.E., Graham, B.W., Chaput, D., and Priddle, M.W. 1985. Retardation of toxic chemicals in a contaminated outwash aquifer. Wat. Sci. Technol. 17:57-69.

Reynolds, W.D., Gillham, R.W. and Cherry, J.A. 1982. Evaluation of distribution coefficients for the prediction of strontium and cesium migration in uniform sand. Canadian Geotechnical Journal, 19(1):92-103.

Reilly, T.E., Franke, O.L. and Bennett, G.D. 1989. Bias in ground water samples caused by wellbore flow. J. Hydraulic Engin., 115(2):270-276.

Reinhard, M., Goodman, N.L. and Barker, J.F. 1984. Occurrence and distribution of organic chemicals in two landfill leachate plumes. Environ. Sci. Technol., 18(12), 953-961.

Robin, M.J.L. and Gillham, R.W. 1987. Field evaluation of well purging procedures. Ground Water Monitoring Review, 7(4):85-93.

Roy, W.R. and Griffin, R.A. 1985. Mobility of organic solvents in water saturated soil materials. Environ. Geol. Wat. Sci., 7(4):241-247.

Sax, N.I. 1984. Dangerous Properties of Industrial Materials. Van Nostrand Reinhold Co., 6th Edition.

Schwarzenbach, R.P. and Westall, J. 1981. Transport of non-polar organic compounds from surface water to groundwater. Laboratory sorption studies. Environ. Sci. Technol. 15(11):1360-1367.

Schwarzenbach, R., Giger, W., Schaffner, C., and Wanner, O. 1985. Groundwater contamination by volatile halogenated alkanes: abiotic formation of volatile sulfur compounds under anaerobic conditions. Environ. Sci. Technol., 19(4):322-327.

Schwille, F. 1975. Groundwater pollution by mineral oil products. In: Proceedings of the Moscow Symposium on Groundwater Pollution, August 1971, IAHS Publ. No. 103, pp. 226-240.

Schwille, F. 1984. Migration of organic fluids immiscible with water in the unsaturated zone. In "Pollutants in Porous Media", Yaron, B., Dagan, G., Goldshmid, J. (Editors), Springer-Verlag, New York, pp. 27-48.

Schwille, F. 1988. "Dense chlorinated solvents in porous and fractured media. Model experiments." Translated by Pankow, J.F., Lewis Publishers, Chelsea, MI.

Semprini, L., Roberts, P.V., Hopkins, G.D. and Mackay, D.M. 1988. A field evaluation of in-situ biodegradation for aquifer restoration. U.S. EPA, Ada, OK, EPA 600/52-87/096.

Simard, G. and Lanctot, J.P. 1987. Decontamination of Ville Mercier aquifer for toxic organics. In: Proceedings of the First International Meeting, NATO/CCMS Pilot Study, Demonstration of remedial action technologies for contaminated land and groundwater, Washington, D.C., 11-13 November 1987, pp. 135-165.

Sitar, N., Hunt, J.R. and Udell, K.S. 1987. Movement of nonaqueous liquids in groundwater. In "Proceedings of the Geotechnical Practice of Waste Disposal '87". Geotechnical Division, A.S.C.E., Ann Arbor, MI, pp. 205-223.

Smith, M.T. 1988. Organic chemicals in ground water: what are the risks and how safe is safe? In: Proceedings of the Sixteenth Biennial Conference on Ground Water, Ed. J.J. DeVries, California Water Resources Centre, University of California, Report No. 66, pp. 139-145.

Srinivasan, P., Patwardhan, V.R., Devotta, S., and Watson, F.A. 1984. Thermal decomposition of trichlorotrifluoroethane (R113) under static and flow conditions. Chem. Eng. Res. Des., 62:266-268.

Stumm, W. and Morgan, J.J. 1981. Aquatic Chemistry. 2nd Edition. Wiley-Interscience, New York, pp. 734-5.

Swallow, K.C., Shifrin, N.S. and Doherty, P.J. 1988. Hazardous organic compound analysis. Environ. Sci. Technol., 22(2):136-142.

Taylor, J.K. 1987. Quality assurance of chemical measurements. Lewis Publishers, Inc., Chelsea, Michigan.

U.S. EPA. 1984a. Method 602 - Purgeable Aromatics. Federal Register, 49(209):40-48.

U.S. EPA. 1984b. Method 624 - Purgeables. Federal Register, 49(209):141-152.

U.S. EPA. 1984c. Method 625 - Base/Neutrals and Acids. Federal Register, 49(209):141-152.

U.S. EPA. 1984d. Method 613 - 2,3,7,8-Tetrachlorodibenzo-p-Dioxin. Federal Register, 49(209):141-152.

U.S. EPA. 1987. Federal Register Part II, U.S. EPA, 40 CFR Part 264 and 270, July 9, 1987.

U.S. EPA. 1988. "Reviews of Environmental Contamination and Toxicology". G.W. Ware, Editor. Springer-Verlag: New York, Vol. 106, pp. 117-120.

Verschueren, K., 1983. Handbook of environmental data on organic chemicals. Van Nostrand Reinhold, New York.

Vigon, B.W. and Rubin, A.J. 1989. Practical considerations in the surfactant-aided mobilization of contaminants in aquifers. Journal WPCF 61(7):1233-1240.

Vogel, T.M. and McCarty, P.L. 1987. Abiotic and biotic transformations of 1,1,1-trichloroethane under methanogenic conditions. Environ. Sci. Technol., 21(12):1208-1213.

Vogel, T.M., Criddle, C.S. and McCarty, P.L. 1987. Transformations of halogenated aliphatic compounds. Environ. Sci. Technol., 21(8):722-736.

WESA (Water and Earth Sciences Associates). 1988. Purge well test. Carp, Ontario.

Whiffin, R.B. and Bahr, J. 1985. Assessment of purge well effectiveness for aquifer decontamination. In: Proc. of the NWWA Conference, 4th National Symposium on Aquifer Restoration and Groundwater Monitoring, published by NWWA, Dublin, OH, pp. 75-81.

Wilson, B.H., Smith, G.B., Rees, J.R. 1986. Biotransformations of selected alkylbenzenes and halogenated aliphatic hydrocarbons in methanogenic aquifer material: a microcosm study. Environ. Sci. Technol., 20(10):997-1002.

Wolf, K., Holland, R. and Rajaratnam, A. 1987. Vinyl chloride contamination: The hidden threat. J. Hazardous Materials, 15:163-184.

Wood, P.R., Lang, R.F. and Payan, I.L. 1985. Anaerobic transformation, transport and removal of volatile chlorinated organics in ground water. In: "Ground Water Quality", edited by C.H. Ward, W. Giger and P.L. McCarty. Wiley-Interscience, New York, Chapter 27.

Zenon, 1986. A study of enhanced dissolution for the in-situ remediation of DNAPL chemicals in the subsurface. Burlington, Ontario.

Table 1. Information concerning five hazardous waste sites in eastern Canada.

Site	Geology	Source of Contamination
Elmira	sequence of sand and gravel aquifers separated by till aquitard	industrial lagoons with chemical process wastes
Gloucester	sequence of sand and gravel aquifers separated by till aquitard	disposal of 100,000 L of lab solvents, wood preservatives and inorganics
Mercier	esker overlying fractured bedrock	disposal of 40,000,000 L of liquid industrial wastes
Smithville	thin till over fractured dolomite	industrial reprocessing of organic liquids
Sydney	thin till over fractured sandstone	coking coal manufacture and coal tar production

Table 2. Concentration ranges of contaminants of concern in ground waters at five hazardous waste sites in eastern Canada. ppm (>1 mg/L), ppb (= 1-1000 µg/L) and ppq (= pg/L) indicate maximum levels detected in several monitoring wells at each site.

Site	DNAPL Composition	Contaminants of Concern
Elmira	presence uncertain	BTEX, phenols and CPs (ppm); 2,4-D and 2,4,5-T (ppb), PCDD (soil)
Gloucester	presence uncertain	F113, solvents, VC, dioxane, benzene (ppb); PCDDs and PCDFs (ppq)
Mercier	oil + PAHs + solvents	solvents, benzene, CB and VC (ppm)
Smithville	oil + TCE + PCB + CBs	PCBs, TCE and CBs (ppb)
Sydney	coal tar	BTEX and napthalene (ppm)

Abbreviations :

BTEX = benzene, toluene, ethyl benzene and xylene
CBs = chlorobenzenes
CPs = chlorophenols
dioxane = 1,4-dioxane
F113 = Freon-113 (1,1,2-trichloro,1,2,2-trifluoroethane)
PCB = polychlorinated biphenyls
PCDD = polychlorinated dibenzodioxins
PCDF = polychlorinated dibenzofurans
solvents = tetrachloroethene (PCE), trichloroethene (TCE), 1,1,2-trichloroethane (TCA), 1,2-dichloroethane
VC = chloroethene (vinyl chloride)
2,4-D = 2,4-dichlorophenoxyacetic acid
2,4,5-T = 2,4,5-trichlorophenoxyacetic acid

Table 3. Drinking water quality guideline values in µg/L (ppb).

Compound	Canadian Recommended Maximum Acceptable Concentration	U.S. EPA Recommended Maximum Contaminant Level
benzene	5	5
chloroethene (VC)	-	2
1,4-dichlorobenzene	5	75
1,2-dichloroethane	-	5
1,1-dichloroethene	-	7
tetrachloromethane	5	5
1,1,1-trichloroethane (TCA)	-	200
trichloroethene (TCE)	-	5

- = not defined
Sources: Guidelines for Canadian Drinking Water Quality, 1987, and Federal Register, 52(130):25690-25717.

Table 4. Ground water chemistry of selected toxic organic chemicals in µg/L at two of the purge wells, Mercier, Quebec, in August 1988, compared to U.S. EPA maximum contaminant levels in drinking water.

Compound	Well 1	Well 3	EPA MCL
Chloroethene (VC)	160	92	2
Benzene	174	162	5
1,2-Dichloroethane	435	723	5
Trichloroethene (TCE)	105	102	5

Sampled and analysed by NWRI
MCLs taken from Federal Register, 52(130), p. 25691, July 8, 1987.

Table 5. Inorganic ground water quality data at selected multilevel samplers (16M, 37M) and piezometers (41P) within the outwash aquifer. See Figures 2 and 4 for location of monitors.

Parameter	Units	41P	16M-16	37M-9
Sp. Cond.[1]	µs/cm	380	730	1380
pH		7.9	7.3	7.2
E_H[2]	volts	0.55	0.13	0.13
O_2	mg/L	2.1	$\leq$0.2	$\leq$0.2
Tit. Alk.[3]	meq/L	3.6	7.0	10.4
TIC[4]	mg/L	47	99	141
TOC[5]	mg/L	1.3	14	61
Ca^{2+}	mg/L	60	120	210
Mg^{2+}	mg/L	21	20	58
Na^{+}	mg/L	1.2	13	88
K^{+}	mg/L	0.9	6.7	3.5
Fe^{2+}	mg/L	0.5	1.5	6.7
Mn^{2+}	mg/L	0.1	n.m.[6]	0.2
SO_4^{2-}	mg/L	32	65	290
Cl^{-}	mg/L	6.2	27	200
SI[7] Calcite		0.34	0.26	0.45
SI Siderite		-6.7	0.16	0.76
SI Ferrihydrite		3.2	1.1	1.4

[1]specific electrical conductance at 10°C
[2]measured Pt electrode potential referred to normal hydrogen electrode
[3]titration alkalinity
[4]total inorganic carbon
[5]total organic carbon
[6]n.m. = not measured
[7]SI = saturation index, log IAP - log K_{sp}

Table 6. E_H and dissolved sulfide in ground waters from the outwash aquifer directly beneath the SWC.

Monitoring Point	E_H (V)	Sulfide (mg/L)
37M-1	0.07	0.01
37M-6	0.06	0.12
37M-7	0.06	0.13
37M-9	0.04	0.075
67M-6	-	0.005
67M-7	-	0.005
67M-9	0.05	1.1
67M-11	-	0.040

See Figure 3 for location of monitoring points
- indicates not measured

Table 7. pH, E_H and dissolved oxygen in ground waters (monitor 37M-9) beneath the SWC in 1982, 1983 and 1988.

	1982	1983	1988
pH	7.2	7.0	7.1
E_H (V)	0.13	0.04	0.11
O_2 (mg/L)	<0.3	<0.3	<0.3

Table 8. Volatile organic compounds present in the outwash aquifer, May 1988.

Compound	Concentration (µg/L)	Frequency[a] %	Aqueous[b] Solubility (mg/L)	% of Aqueous[f] Solubility
Dioxane	~300-2000	13	-[e]	
Diethyl ether	<2-658	68	84300(15)[c]	0.0008
Tetrahydrofuran	20-230	32	-[e]	
Benzene	0.9-646	65	1780(20)[c]	0.04
Toluene	<0.5-2.5	30	470(16)[c]	0.0005
Ethylbenzene	2	3	140(15)[c]	0.001
o-Xylene	10	3	175(20)[c]	0.006
2-Ethyltoluene	2.1	3	-[d]	
Dichloromethane	4-60	16	15938	0.0004
Trichloromethane	<2-164	51	8983	0.002
Tetrachloromethane	<2-7.5	11	683	0.001
Chloroethane	<6	3	5742	0.0001
Chloroethene	<1-40	13	2763(25)	0.0015
1,1-Dichloroethane	<1-614	54	5193	0.012
1,1,-Dichloroethene	0.9-60	43	6650	0.0009
1,2-Dichloroethane	3.9-58	30	8579	0.0007
Trans-1,2-dichloroethene	<1-2.8	16	6260(25)	0.00004
Cis-1,2-dichloroethene	<1-7.5	19	4000	0.0002
1,1,1-Trichloroethane	<3-520	43	1707	0.03
1,1,2-Trichloroethane	<2	3	4564	0.00004
Trichloroethene	7.4-583	41	1061	0.055
Tetrachloroethane	5-15	8	763	0.002
Tetrachloroethene	2-105	27	149	0.07
1,2-Dichloropropane	3.2-11	5	2300	0.0005
Chlorobenzene	1.1-134	11	193	0.069
1,1,2-Trichloro-1,2,2-trifluoroethane	<5-2725	35	136	2

[a]Frequency of identification of compounds in 37 samples collected from the network shown in Figure 3.
[b]Solubilities from Horvath (1982) at 10°C unless otherwise indicated in parentheses.
[c]Solubility from Verschueren (1983), temperature in parentheses.
[d]No data available.
[e]Miscible.
[f]Value indicates the percentage of aqueous solubility represented by the maximum measured concentration.

Table 9. Volatile organic compounds tentatively identified by computerized library searches. All concentrations estimated to be <50 µg/L.

1,2-Dichloro-1,2,2-trifluoroethane
1,1,2-Trichloro-1,2-difluoroethane
Dichlorodifluoromethane
Chlorotrifluoroethene
Diisopropyl ether
Pentane
Methoxybenzene
Diethyl sulfide
Dimethyl sulfide
Methylethyl sulfide
Tetrahydrothiophene
Chloropropene
1,2-Dibromomethene

Table 10. Volatile organic compounds (in µg/L) identified in monitoring well network, May 1988.

Sample	124P	125P	126P	127P	128P
Dioxane				≈330	
Diethyl ether	643	72	331	122	
Tetrahydrofuran	70		230	180	
Benzene	2.8	120			
Toluene		3			
Ethylbenzene		3			
m+p-Xylene		2.6			
o-Xylene		10			
Trichloromethane		52			<2
Tetrachloromethane					<4
Chloroethane	<1				
1,1-Dichloroethane	2.6	24			0.9
1,2-Dichloroethane	10.3	14			
1,1-Dichloroethene		155			<2
Trans-1,2-dichloroethene					<1
Cis-1,2-Dichloroethene					<1
1,1,1-Trichloroethane		193			<3
Trichloroethene		505			
Tetrachloroethene		60			
1,2-Dichloropropane		12			
Chlorobenzene		68			
1,1,2-Trichloro-1,2,2-trifluoroethane		200			

Blank spaces indicate compound not detected.

Table 11. Product distribution expressed as percentage of total chlorinated ethanes/ethenes found in 37M, 67M and 92M.

Compound	1983	1988
TCA	27.3	27.6
1,1-DCA	39.5	25.4
1,1-DCE	1.4	5.1
PCE	9.7	3.3
TCE	22.0	36.8
CIS	n.a.	0.5
TRANS	0.1	0.2
VC	n.a.	1.0

n.a. Not available

Table 12. Retardation factors.

Chemical	log K_{ow}	R_{f1}	R_{f2}	R_{f3}	R_{f4}	R_{f5}	R_{f6}	R_{f7}
DIOXANE	-0.27	1.6	1.4	1.6	1.0	1.1	1.2	1.4(2)
THF	0.46	2.2	2.2	2.5	1.0			
DEE	0.89	3.3	3.0	3.4	1.1	1.7	1.6	2.2(2) 3.2(4)
12DCA	1.48	7.6		5.7	1.2	7.2	5.0	13.1(4)
TCM	1.97			9.3	1.3	7.5	5.0	14.9(4)
BENZENE	2.04	8.8		10.0	1.4	14.3	8.0	30.0(4)
11DCE	2.13			11.0	1.5	10.7	7.0	18.8(4)

RETARDATION FACTORS BASED ON:

1. Field data (length of organic plume vs. length of chloride plume)
2. Purge well evaluation test
3. Correlation log (R_f-1) = 0.5*log K_{ow} - 0.065 derived from field data (Patterson et al., 1985)
4. Equation $R_f=1+(\text{bulk density/porosity})*K_d$ where $K_d = 3.2*f_{oc}*(K_{ow})*\exp 0.72$ (Schwarzenbach and Westall, 1981)
5. Column test #3 (centre of mass of organic vs. centre of mass of iodide)
6. Column test #3 (C/Cmax = 0.5 of organic vs. C/Cmax = 0.5 of iodide)
7. Column test #3 (CXTFIT: number in parenthesis refers to Model 2 (LEA) or 4 (nonequilibrium))

NOTES:

Bulk density - 1.76 g cm^{-3}
Porosity - 0.32
foc - 0.00075

Table 13. Corrective action alternatives (modified from OTA, 1984).

1. MANAGEMENT OPTIONS

   - limit/terminate aquifer use
   - develop alternative water supply
   - remove source of contamination
   - monitoring
   - no action

2. CONTAINMENT

   - physical barriers, e.g., slurry walls, liners, clay, caps, etc.
   - hydraulic control via pumping and/or recharging

3. WITHDRAWAL (AND TREATMENT)

   - pumping
   - excavation
   - gravity drainage
   - gas venting

4. IN-SITU RESTORATION

   - biodegradation
   - steam injection
   - surfactant injection
   - enhanced solubilization

Table 14. Volatile organic compounds measured by GC/MS in µg/L in ground waters from selected multilevel samplers within the outwash aquifer. See Figure 6 for locations.

Compound	16M-16		37M-9	
	1982	1988	1982	1988
Methylene chloride	620	30	6,050	25
Chloroform	12,900	129	32,000	24
Bromoform	90	n.d.	n.d.	n.d.
Vinyl Chloride	-	n.d.	-	40
1,1-Dichloroethene	20	n.d.	30	60
1,2-Dichloroethane	480	3.9	2,700	25
1,1,1-Trichloroethane	1,900	141	3,500	176
Trichloroethene	1,600	38	20	110
Tetrachloroethene	4,500	62	n.d.	n.d.
Benzene	580	6	17,000	646

NOTES:

- means not measured
n.d. means not detectable

APPENDIX

COMPLETE 1984 AND 1988 DATA FOR VOLATILE ORGANIC CHEMICALS IN THE OUTWASH AQUIFER, GLOUCESTER LANDFILL

TABLE 1. 1984 VOLATILE ORGANIC ANALYSIS

(BY D. CHAPUT AND R.J. PATTERSON, NHRI)

Table 1. 1984 Volatile Organic Analysis

Sampling Point	DEE ug/L	11-DCE ug/L	MECL ug/L	T12-DE ug/L	11-DCA ug/L	ACET ug/L	CHCL3 ug/L	TEP ug/L	111-TCA ug/L	12-DCA ug/L	BENZ ug/L	CCL4 ug/L	TCE ug/L	14DWE ug/L	TOL ug/L	PCE ug/L	CLBEN ug/L	ETBEN ug/L	CHBR3 ug/L
33H1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33H2	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33H3	0.6	0.0	0.0	0.0	0.0	0.0	0.0	23.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33H6	156.3	0.0	0.0	0.0	0.0	0.0	0.0	15.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33H8	249.4	0.0	0.0	0.0	0.0	0.0	0.0	47.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33H9	26.7	0.0	0.0	0.0	0.0	0.0	0.0	46.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33H10	17.1	0.0	0.0	0.0	0.0	0.0	0.0	24.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33H11	9.3	0.0	0.0	0.0	0.0	0.0	0.0	4.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33H12	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33H13	26.4	0.0	0.0	0.0	0.0	0.0	0.0	12.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33H14	3.2	0.0	0.0	0.0	0.0	0.0	0.0	3.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33H15	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
33H16	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
34H2	368.7	0.0	0.0	0.0	0.0	0.0	0.0	352.8	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34H3	878.3	0.0	0.0	0.0	0.0	0.0	0.0	669.0	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34H4	754.5	0.0	0.0	0.0	0.0	0.0	0.0	288.7	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34H5	951.4	0.0	0.0	0.0	0.0	0.0	0.0	414.5	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34H6	3.2	0.0	0.0	0.0	0.0	0.0	0.0	277.3	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34H7	2.1	0.0	0.0	0.0	0.0	0.0	0.0	3.9	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34H8	1.0	0.0	0.0	0.0	0.0	0.0	0.0	4.2	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34H9	2.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
34H11	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	P	0.4	0.0	0.0	0.0	0.0
34H12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	P	0.5	0.0	0.0	0.0	0.0
34H16	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0
37H1	0.0	4.6	3.4	0.0	5.5	0.0	31.7	0.0	7.6	1.6	96.5	4.6	22.2	0.0	1.2	1.2	0.1	0.0	0.0
37H2	2.0	5.7	3.0	0.0	4.8	0.0	27.0	0.0	5.0	1.1	91.5	3.5	17.1	0.0	1.0	0.9	0.0	0.1	0.0
37H3	4.4	3.5	1.6	0.0	3.3	0.0	16.5	7.9	4.4	1.0	60.2	2.4	13.2	0.0	1.0	1.0	0.0	0.2	0.0
37H4	111.2	<5.00	0.6	0.0	15.1	0.0	12.0	19.2	2.5	18.6	43.1	1.9	10.4	0.0	0.7	0.8	0.0	0.1	0.0
37H5	292.6	<5.00	0.5	0.0	38.6	0.0	14.1	130.5	2.2	36.0	320.2	2.2	11.0	0.0	1.2	0.6	0.0	0.1	0.0
37H6	191.0	<5.00	1.9	0.0	58.0	0.0	18.3	48.2	1.9	35.3	536.5	2.0	11.2	0.0	1.7	0.8	0.0	0.6	0.0
37H7	406.5	8.5	3.6	0.0	106.6	0.0	14.4	235.2	0.9	96.9	988.8	2.3	9.4	0.0	4.6	0.7	0.0	0.7	0.0
37H8	850.7	27.5	440.0	0.0	662.0	0.0	7265.7	865.4	10.5	555.8	500.0	10.2	14.9	0.0	2.6	1.9	0.0	0.8	0.0
37H9	979.5	144.0	860.0	0.0	1416.3	0.0	8632.9	874.5	158.8	822.5	3394.8	14.9	36.7	0.0	4.7	3.6	0.0	1.6	0.0
37H10	558.5	793.0	754.0	0.0	1768.3	0.0	9645.9	485.8	1182.0	847.9	3182.8	6.9	220.2	0.0	8.2	3.3	0.0	1.1	0.0
37H11	373.5	1093.0	578.0	0.0	1591.4	0.0	4556.5	164.3	2234.8	475.3	2312.3	15.2	697.2	0.0	19.6	0.0	0.0	1.4	0.0
37H12	164.2	747.0	150.0	0.0	1571.0	0.0	723.5	0.0	1096.0	201.0	1690.4	8.6	1525.1	0.0	35.5	0.0	0.0	0.0	0.0
37H13	138.8	785.0	100.0	0.0	1257.1	0.0	335.6	0.0	802.7	134.1	1473.8	8.1	1616.7	0.0	585.9	0.0	0.0	0.0	0.0
37H14	124.9	530.0	T	0.0	673.1	0.0	97.0	0.0	652.6	99.8	1562.1	5.0	371.6	0.0	10.7	0.0	0.0	0.0	0.0
37H15	68.3	80.0	5.0	8.7	98.1	0.0	144.9	0.0	15.2	59.1	514.2	0.0	17.4	0.0	4.0	0.0	0.0	0.0	0.0
37H16	19.1	22.4	14.5	0.0	10.0	0.0	423.0	0.0	34.7	18.0	48.9	123.4	169.7	0.0	1.4	14.2	2.8	0.0	143.3
37H17	18.4	12.2	12.7	0.0	8.0	0.0	358.3	0.0	32.9	18.8	35.3	128.7	113.0	0.0	3.2	13.1	3.3	0.0	95.0
37H18	22.9	11.3	17.3	0.0	6.6	0.0	378.3	15.5	33.0	21.5	38.7	170.0	130.6	0.0	1.6	19.7	4.5	0.0	120.2
37H19	26.3	9.1	19.6	0.0	5.4	0.0	359.2	13.7	26.2	19.3	30.5	179.4	134.2	0.0	0.0	16.1	3.2	0.0	140.0
38H2	45.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.8	2.6	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
38H3	23.8	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	7.9	2.3	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0
38H4	162.2	0.0	0.0	0.0	0.4	0.0	0.0	80.0	0.0	16.4	2.6	0.0	0.3	P	0.0	0.0	0.0	0.0	0.0
38H6	308.4	0.0	0.0	0.0	0.4	0.0	0.0	22.1	0.0	11.1	2.5	0.0	0.4	P	0.0	0.0	0.0	0.0	0.0
38H8	80.1	0.0	0.0	0.0	0.3	0.0	0.0	2.2	0.0	4.8	0.2	0.0	0.2	P	0.0	0.0	0.0	0.0	0.0

Table 1. 1984 Volatile Organic Analysis

Sampling Point	DEE ug/L	11-DCE ug/L	MECL ug/L	T12-DE ug/L	11-DCA ug/L	ACET ug/L	CHCL3 ug/L	TRF ug/L	111-TCA ug/L	12-DCA ug/L	BENZ ug/L	CCL4 ug/L	TCE ug/L	14DBE ug/L	TOL ug/L	PCE ug/L	CLBEN ug/L	ETBEN ug/L	CHBR3 ug/L
38N9	160.9	0.0	0.0	0.0	0.2	0.0	0.0	2.8	0.0	1.7	1.2	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
38N11	112.8	0.0	0.0	0.0	0.5	0.0	0.0	120.5	0.0	1.7	12.9	0.0	3.4	P	0.0	0.0	0.0	0.0	0.0
38N13	455.5	0.0	0.0	0.0	2.3	0.0	4.5	215.5	0.6	3.7	28.7	0.0	18.8	P	0.3	0.0	0.0	0.0	0.0
38N15	568.8	0.0	0.0	0.0	23.4	0.0	0.0	445.4	0.0	26.8	122.1	0.0	6.7	P	0.0	0.0	0.0	0.0	0.0
38N17	501.9	0.0	0.0	0.0	13.2	0.0	0.0	180.7	0.0	9.4	62.0	0.0	3.0	P	0.0	0.0	0.0	0.0	0.0
38N18	21.1	0.0	0.0	0.0	0.5	0.0	0.5	3.3	0.0	2.5	4.0	0.0	1.4	0.0	0.0	0.0	0.0	0.0	0.0
56N4	71.6	0.0	1.0	0.0	7.6	0.0	1.1	268.7	0.0	19.5	110.1	0.0	12.9	0.0	2.5	0.6	6.4	41.2	0.0
56N7	1.9	0.0	0.3	0.0	6.2	0.0	1.0	23.7	0.0	0.0	6.1	0.0	5.0	0.0	12.8	0.0	0.0	6.9	0.0
67N1	0.0	2.1	0.0	0.0	0.0		5.2		5.1	0.0	10.7	0.0	38.0	0.0	0.0	0.0	0.0	0.0	0.0
67N3	0.0	2.1	0.0	0.0	2.0		1.8		1.5	2.5	38.7	0.0	24.5	0.0	0.0	0.0	0.0	0.0	0.0
67N4	0.0	2.6	0.0	0.0	3.4		11.8		17.7	3.3	45.1	0.0	62.3	0.0	0.0	0.0	0.0	0.0	0.0
67N5	0.0	1.2	0.0	0.0	2.2		1.9		1.2	1.1	31.0	0.0	31.4	0.0	0.0	0.0	0.0	0.0	0.0
67N6	0.0	1.4	0.0	0.0	14.4		4.1		11.2	8.5	154.0	0.0	41.9	0.0	0.0	0.0	0.0	0.0	0.0
67N7	0.0	1.4	0.0	0.0	9.8		1.1		12.1	4.5	90.4	0.0	38.4	0.0	0.0	0.0	0.0	0.0	0.0
67N8	0.0	0.0	0.0	0.0	1.5		0.0		1.0	0.7	27.3	0.0	23.5	0.0	0.0	0.0	0.0	0.0	0.0
67N9	79.0	0.0	0.0	0.0	155.0		15.2		283.0	50.3	418.0	0.0	854.0	0.0	0.0	0.0	0.0	0.0	0.0
67N10	188.0	P			234.0		131.8		300.0	105.0	372.0	0.0	694.0	0.0	0.0	0.0	0.0	0.0	0.0
67N11	190.0	P	?	0.0	112.0		2885.0		645.0	>200.00	289.0	0.0	1561.0	0.0	0.0	0.0	0.0	0.0	0.0
67N12	62.0		0.0	0.0	69.0		1754.0		400.0	200.0	228.0	0.0	742.0	0.0	0.0	0.0	0.0	0.0	0.0
67N14	35.0		0.0	0.0	63.0		171.0		69.0	30.0	192.0	0.0	253.0	0.0	0.0	0.0	0.0	0.0	0.0
67N15	14.0		0.0	0.0	14.0		188.0		16.0	11.0	96.0	0.0	62.0	0.0	0.0	0.0	0.0	0.0	0.0
67N16	11.0		0.0	0.0	25.0		324.0		65.0	14.0	79.0	0.0	162.0	0.0	0.0	0.0	0.0	0.0	0.0
67N17	5.7		0.0	0.0	22.0		470.0		86.0	22.0	36.0	0.0	149.0	0.0	0.0	0.0	0.0	0.0	0.0
67N18	10.0		0.0	0.0	23.0		5.0		89.0	26.0	52.0	0.0	181.0	0.0	0.0	0.0	0.0	0.0	0.0
67N19	17.0		0.0	0.0	97.0		803.0		183.0	44.0	123.0	0.0	272.0	0.0	0.0	0.0	0.0	0.0	0.0
67N20	7.0		0.0	0.0	23.0		452.0		100.0	18.0	33.0	15.0	170.0	0.0	0.0	0.0	0.0	0.0	0.0
67N21	0.0		0.0	0.0	8.7		894.0		202.0	18.0	T	29.0	36.0	0.0	0.0	0.0	0.0	0.0	0.0
67N22	0.0		0.0	0.0	2.7		101.0		59.0	<2.00	0.0	7.0	18.0	0.0	0.0	0.0	0.0	0.0	0.0
67N23	0.0		0.0	0.0	1.0		22.0		21.0	<2.00	T	T	11.0	0.0	0.0	0.0	0.0	0.0	0.0
67N24	0.0		0.0	0.0	0.0		37.0		38.0	0.0	22.0	0.0	72.0	0.0	0.0	0.0	0.0	0.0	0.0
92N1	0.7	0.0	0.0	0.0	0.0	0.0	1.0	0.0	1.0	0.3	1.7	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0
92N2	1.7	1.0	0.0	0.0	5.2	0.0	1.4	0.0	3.7	1.1	9.4	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0
92N3	4.1	2.1	0.0	0.0	11.1	0.0	2.4	0.0	4.7	2.2	11.1	0.0	2.2	0.0	0.0	0.0	0.0	0.0	0.0
92N5	13.1	35.5	0.0	0.0	32.3	0.0	4.3	0.0	33.8	10.0	37.3	0.0	9.1	0.0	0.0	0.0	0.0	0.0	0.0
92N6	0.0	41.5	0.0	1.4	35.5	0.0	36.0	5.9	106.4	9.9	74.6	0 0	46.5	0.0	0.0	0.0	0.0	0.0	0.0
92N7	15.6	31.2	0.0	0.0	25.6	0.0	40.7	30.9	77.3	8.6	71.1	0.0	43.6	0.0	0.0	0.0	0.0	0.0	0.0
92N9	48.3	245.9	0.0	0.0	176.4	0.0	105.9	55.0	253.3	53.8	211.1	0.0	154.8	0.0	0.0	0.0	0.0	0.0	0.0
92N11	16.0	15.0	0.0	0.0	15.5	0.0	42.3	0.0	35.4	10.3	38.3	0.0	22.5	0.0	0.0	0.0	0.0	0.0	0.0
92N13	11.0	73.9	0.0	0.0	18.0	0.0	472.2	0.0	144.1	27.4	74.5	0.0	52.8	0.0	0.0	0.0	0.0	0.0	0.0
92N14	15.6	34.4	0.0	1.8	18.9	0.0	272.7	0.0	75.5	19.9	44.7	0.0	39.0	0.0	0.0	0.0	0.0	0.0	0.0
92N15	9.3	38.9	0.0	0.9	15.0	0.0	310.1	10.5	87.8	18.2	42.5	0.0	46.1	0.0	0.0	0.0	0.0	0.0	0.0
92N17	0.0	66.4	0.0	0.0	28.2	0.0	179.4	0.0	190.8	0.0	26.3	0.0	30.7	0.0	0.0	0.0	0.0	0.0	0.0
93N15	713.9	130.9	5.5	0.0	59.3	0.0	48.1	68.5	188.2	31.2	164.4	0.0	54.0	0.0	1.6	0.0	0.0	0.0	0.0
93N17	484.7	82.2	5.5	0.0	130.1	0.0	147.8	90.7	287.0	64.7	387.0	0.0	77.0	0.0	0.0	0.0	0.0	0.0	0.0
93N19	26.6	5.9	5.2	0.0	26.1	0.0	26.7	18.1	22.3	22.5	164.5	0.0	5.2	0.0	0.0	0.0	0.0	0.0	0.0
93N20	11.9	6.5	1.8	0.0	8.7	0.0	9.2	0.0	14.3	6.7	59.7	0.0	6.5	0.0	0.0	0.0	0.0	0.0	0.0
93N21	8.2	7.3	1.0	0.0	7.0	0.0	0.0	0.0	12.3	5.7	43.7	0.0	3.4	0.0	0.0	0.0	0.0	0.0	0.0

Table 1. 1984 Volatile Organic Analysis

Sampling Point	DEE ug/L	11-DCE ug/L	MECL ug/L	T12-DE ug/L	11-DCA ug/L	ACET ug/L	CHCL3 ug/L	THF ug/L	111-TCA ug/L	12-DCA ug/L	BENZ ug/L	CCL4 ug/L	TCE ug/L	14DXE ug/L	TOL ug/L	PCE ug/L	CLBEN ug/L	ETBEN ug/L	CXBE3 ug/L
93N23	3.6	12.0	0.5	0.0	12.9	0.0	1.9	0.0	25.7	0.0	4.2	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
94N2	252.9	0.0	0.0	0.0	0.0	0.0	0.0	49.9	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94N5	1000.0	0.0	0.0	0.0	0.0	0.0	0.5	96.8	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94N6	641.6	0.0	0.0	0.0	0.0	0.0	0.3	229.4	0.2	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94N9	1254.4	0.0	0.0	0.0	0.0	0.0	0.0	573.0	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94N10	974.4	0.0	0.0	0.0	0.0	0.0	0.0	819.7	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94N11	1022.8	0.0	0.0	0.0	0.0	0.0	0.0	799.0	0.0	0.0	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94N12	1152.5	0.0	0.0	0.0	0.0	0.0	1.0	1050.2	0.0	0.3	0.0	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94N18	1113.4	0.0	0.0	0.0	14.0	0.0	0.0	210.5	2.0	9.7	159.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
94N21	747.5	0.0	0.0	0.0	0.0	0.0	0.0	171.0	0.0	0.8	8.3	0.0	0.0	P	0.0	0.0	0.0	0.0	0.0
94N22	89.4	0.0	0.0	0.0	0.3	0.0	0.0	5.6	0.0	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
94N23	1.3	0.0	0.0	0.0	0.8	0.0	0.4	0.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
94N24	10.0	0.0	0.0	0.0	1.1	0.0	0.0	0.0	0.9	0.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95N1	1.1	0.0	0.0	0.0	0.0	0.0	2.7	0.0	0.0	0.5	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95N2	0.7	1.1	0.0	0.0	0.5	0.0	0.0	0.0	0.0	1.0	0.8	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0
95N3	35.1	0.4	0.0	0.0	0.4	0.0	2.5	4.9	0.8	0.0	0.2	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
95N5	254.8	36.7	0.0	0.0	22.4	0.0	4.3	104.6	4.0	3.3	3.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95N6	232.2	2.5	1.5	0.0	36.9	0.0	41.8	133.2	2.1	6.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95N7	201.1	3.4	3.7	0.0	58.8	0.0	108.9	90.1	4.6	6.9	4.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95N9	211.0	66.7	11.1	0.0	99.5	0.0	177.4	114.1	149.5	18.5	34.3	0.0	3.2	0.0	0.0	0.0	0.0	0.0	0.0
95N11	91.3	56.6	11.1	0.0	53.0	0.0	164.6	10.0	152.0	15.9	41.4	0.0	3.9	0.0	0.0	0.0	0.0	0.0	0.0
95N13	17.8	44.9	44.4	0.0	33.9	0.0	158.0	0.0	125.5	13.8	43.1	0.0	4.2	0.0	0.0	0.0	0.0	0.0	0.0
95N14	22.2	74.7	55.6	0.0	38.9	0.0	589.4	0.0	186.7	27.4	68.5	0.0	7.6	0.0	0.0	0.0	0.0	0.0	0.0
95N15	36.1	100.0	100.0	0.0	39.7	0.0	1107.9	46.3	356.6	100.0	116.0	0.0	20.9	0.0	0.0	0.0	0.0	0.0	0.0
95N17	22.2	124.2	41.1	0.0	36.1	0.0	669.1	0.0	270.7	77.8	68.7	0.0	4.9	0.0	0.0	0.0	0.0	0.0	0.0
95N18	26.4	57.2	23.4	0.0	62.7	0.0	381.1	0.0	176.7	18.6	45.0	0.0	3.3	0.0	0.0	0.0	0.0	0.0	0.0
95N19	22.2	123.8	88.9	0.0	38.4	0.0	614.8	0.0	308.2	32.2	66.0	0.0	8.7	0.0	0.0	0.0	0.0	0.0	0.0
95N20	18.4	38.7	14.7	0.0	73.3	0.0	266.5	0.0	107.8	14.2	34.1	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0
95N21	12.6	38.2	12.2	0.0	97.8	0.0	117.5	0.0	68.6	12.3	31.4	0.0	2.3	0.0	0.0	0.0	0.0	0.0	0.0
95N22	38.2	3.3	5.5	0.0	76.2	0.0	22.8	0.0	16.5	6.7	12.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95N23	25.9	2.2	3.2	0.0	36.4	0.0	16.6	19.8	11.6	4.3	7.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
95N24	38.5	2.2	1.8	0.0	15.9	0.0	15.0	0.0	8.9	2.3	3.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

TABLE 2. 1988 VOLATILE ORGANIC ANALYSIS

(BY S. LESAGE, M.W. PRIDDLE AND P. RIEMANN, NWRI)

NATIONAL WATER RESEARCH INSTITUTE
GROUNDWATER CHEMISTRY LABORATORY
GC-MS ANALYSIS REPORT

DATE: 89/07/12

SAMPLE NUMBER	88GL124P	88GL126P	88GL127P	88GL128P	88GL12N14	88GL14BL2	88GL14N02
CONCENTRATION	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
CHLOROMETHANE							
BROMOMETHANE							
CHLOROETHANE							
VINYL CHLORIDE	<1						
1,1-DICHLOROETHENE				<2			
DICHLOROMETHANE							
DIETHYL ETHER	643	331	122				
TETRAHYDROFURAN	70	230	180				30
TRANS-1,2-DICHLOROETHENE				<1			
1,1-DICHLOROETHANE	2.6			0.9			<3
CIS-1,2-DICHLOROETHENE				<1			
CHLOROFORM				<2			
1,1,1-TRICHLOROETHANE				<3			
CARBON TETRACHLORIDE				<4			
BENZENE	2.8					15	5.9
1,2-DICHLOROETHANE	10.3						
TRICHLOROETHYLENE							
1,2-DICHLOROPROPANE							
DIBROMOMETHANE							
TRANS-1,3-DICHLOROPROPENE							
TOLUENE						16	<0.5
CIS-1,3-DICHLOROPROPENE							
1,1,2-TRICHLOROETHANE							
TETRACHLOROETHENE							
DIBROMOCHLOROMETHANE							
DIBROMOETHANE							
CHLOROBENZENE							
ETHYL BENZENE						6.2	
m+p-XYLENE							
o-XYLENE						2.7	
STYRENE							
BROMOFORM							
CUMENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE							
2-ETHYLTOLUENE							
TRIMETHYL BENZENE							
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BROMOCHLOROMETHANE	93	87	88	72	94	93	91

* SURROGATE was spiked at 100 ug/L

NATIONAL WATER RESEARCH INSTITUTE
GROUNDWATER CHEMISTRY LABORATORY
GC-MS ANALYSIS REPORT

DATE: 89/07/12

SAMPLE NUMBER	89GL14N03	89GL16N04	89GL16N08	89GL16N12	89GL16N16	89GL16NBL	89GL20N01
CONCENTRATION	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
CHLOROMETHANE							
BROMOMETHANE							
CHLOROETHANE							
VINYL CHLORIDE							
1,1-DICHLOROETHENE				48		4.4	
DICHLOROMETHANE					30		
DIETHYL ETHER		<2	<3	<6	10		<3
TETRAHYDROFURAN	60				15		
TRANS-1,2-DICHLOROETHENE							
1,1-DICHLOROETHANE				6.6	2.9	5.5	
CIS-1,2-DICHLOROETHENE							
CHLOROFORM		4.2		27	129	4.9	<2
1,1,1-TRICHLOROETHANE				76	141	5.1	<3
CARBON TETRACHLORIDE							<2
BENZENE				12	6	6	
1,2-DICHLOROETHANE				6.6	3.9		
TRICHLOROETHYLENE				60	38		
1,2-DICHLOROPROPANE							
DIBROMOMETHANE							
TRANS-1,3-DICHLOROPROPENE							
TOLUENE	<0.8					18	
CIS-1,3-DICHLOROPROPENE							
1,1,2-TRICHLOROETHANE							
TETRACHLOROETHENE					62		
DIBROMOCHLOROMETHANE							
DIBROMOETHANE							
CHLOROBENZENE							
ETHYL BENZENE						1.6	
m+p-XYLENE						10	
o-XYLENE						3	
STYRENE							
BROMOFORM							
CUMENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE							
2-ETHYLTOLUENE					2.1		
TRIMETHYL BENZENE						1.6	
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BROMOCHLOROMETHANE	90	95	99	96	83	95	93

* SURROGATE was spiked at 100 ug/L

NATIONAL WATER RESEARCH INSTITUTE
GROUNDWATER CHEMISTRY LABORATORY
GC-MS ANALYSIS REPORT

DATE: 89/07/12

SAMPLE NUMBER	88GL20N10	88GL21N01	88GL21N02	88GL21N03	88GL21NBL	88GL37BL	88GL37N01
CONCENTRATION	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
CHLOROMETHANE							
BROMOMETHANE							
CHLOROETHANE							
VINYL CHLORIDE	2.0						
1,1-DICHLOROETHENE						4	10
DICHLOROMETHANE							
DIETHYL ETHER	70	36	650	616	22		
TETRAHYDROFURAN			120	100			
TRANS-1,2-DICHLOROETHENE							<5
1,1-DICHLOROETHANE						32	4.7
CIS-1,2-DICHLOROETHENE							<5
CHLOROFORM							6.6
1,1,1-TRICHLOROETHANE						5.6	15
CARBON TETRACHLORIDE							7.2
BENZENE	3.0	<1	5	<1	5.7	54	13
1,2-DICHLOROETHANE						8.2	
TRICHLOROETHYLENE						5.4	54
1,2-DICHLOROPROPANE							
DIBROMOMETHANE							
TRANS-1,3-DICHLOROPROPENE							
TOLUENE	1.7	<0.5	<0.5		17	13	0.8
CIS-1,3-DICHLOROPROPENE							
1,1,2-TRICHLOROETHANE							
TETRACHLOROETHENE							3.5
DIBROMOCHLOROMETHANE							
DIBROMOETHANE							
CHLOROBENZENE							1.1
ETHYL BENZENE					1.3	0.3	
m+p-XYLENE					9	10.3	
o-XYLENE					2.8	3.3	
STYRENE							
BROMOFORM							
CUMENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE						1.5	
2-ETHYLTOLUENE						0.3	
TRIMETHYL BENZENE					1.4	1.8	
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BROMOCHLOROMETHANE	92	93	95	92	97	80	85

* SURROGATE was spiked at 100 ug/L

NATIONAL WATER RESEARCH INSTITUTE
GROUNDWATER CHEMISTRY LABORATORY
GC-MS ANALYSIS REPORT

DATE: 89/07/12

SAMPLE NUMBER	89GL37N06	89GL37N07	89GL37N09	89GL47N01	89GL47N02	89GL47N04	89GL54BL
CONCENTRATION	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
CHLORONETHANE							
BRONONETHANE							
CHLOROETHANE							
VINYL CHLORIDE			=40	<1			
1,1-DICHLOROETHENE	35.	3.8	60	<2			
DICHLORONETHANE			25				
DIETHYL ETHER	41	172	254				22
TETRAHYDROFURAN							
TRANS-1,2-DICHLOROETHENE				<2			
1,1-DICHLOROETHANE	12	25	614	2.6	<3		
CIS-1,2-DICHLOROETHENE			7.5	<2			
CHLOROFORN	2.1		24			<2	
1,1,1-TRICHLOROETHANE			170	<4			
CARBON TETRACHLORIDE							
BENZENE	41	118	646		10.5	2.6	5.7
1,2-DICHLOROETHANE	5.2	14	25				
TRICHLOROETHYLENE	24	7.4	110				
1,2-DICHLOROPROPANE			11				
DIBRONONETHANE							
TRANS-1,3-DICHLOROPROPENE							
TOLUENE			2.5		<0.5	<0.3	18
CIS-1,3-DICHLOROPROPENE							
1,1,2-TRICHLOROETHANE							
TETRACHLOROETHENE	3.0	2.0					
DIBRONOCHLORONETHANE							
DIBRONOETHANE							
CHLOROBENZENE							
ETHYL BENZENE							2.2
m+p-XYLENE							9.4
o-XYLENE							2.7
STYRENE							
BRONOFORN							
CUNENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE							
2-ETHYLTOLUENE							
TRINETHYL BENZENE							1.5
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BRONOCHLORONETHANE	37	86	77	120	77	128	96

* SURROGATE was spiked at 100 ug/L

NATIONAL WATER RESEARCH INSTITUTE
GROUNDWATER CHEMISTRY LABORATORY
GC-MS ANALYSIS REPORT

DATE: 89/07/12

SAMPLE NUMBER	89GL54N06	89GL54N08	89GL54N10	89GL67N01	89GL67N06	89GL67N07	89GL67N08
CONCENTRATION	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
CHLOROMETHANE							
BROMOMETHANE							
CHLOROETHANE							
VINYL CHLORIDE							
1,1-DICHLOROETHENE				1.2	0.9		
DICHLOROMETHANE							
DIETHYL ETHER	250	269	258				
TETRAHYDROFURAN	30	45	20				
TRANS-1,2-DICHLOROETHENE							
1,1-DICHLOROETHANE				2.1	3.1		
CIS-1,2-DICHLOROETHENE							
CHLOROFORM				2.0	2.0	2.5	1.0
1,1,1-TRICHLOROETHANE				7	7		
CARBON TETRACHLORIDE							
BENZENE		3.6		0.8	21	4.5	
1,2-DICHLOROETHANE							
TRICHLOROETHYLENE				12	27	32	30
1,2-DICHLOROPROPANE							
DIBROMOMETHANE							
TRANS-1,3-DICHLOROPROPENE							
TOLUENE							
CIS-1,3-DICHLOROPROPENE							
1,1,2-TRICHLOROETHANE							
TETRACHLOROETHENE				3.6	3.0	<2.5	<2.5
DIBROMOCHLOROMETHANE							
DIBROMOETHANE							
CHLOROBENZENE							
ETHYL BENZENE							
m+p-XYLENE							
o-XYLENE							
STYRENE							
BROMOFORM							
CUMENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE							
2-ETHYLTOLUENE							
TRIMETHYL BENZENE							
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BROMOCHLOROMETHANE	95	96	97	95	87	80	97

* SURROGATE was spiked at 100 ug/L

NATIONAL WATER RESEARCH INSTITUTE
GROUNDWATER CHEMISTRY LABORATORY
GC-MS ANALYSIS REPORT

DATE: 89/07/12

SAMPLE NUMBER	88GL67N09	88GL67N11	88GL82N06	88GL82N08	88GL92N15	88GL92N17	88GL92N19
CONCENTRATION	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
CHLOROMETHANE							<8
BROMOMETHANE							<1
CHLOROETHANE							16
VINYL CHLORIDE	4.5	37			7.3	18	4.1
1,1-DICHLOROETHENE		81			7.3	11	4.3
DICHLOROMETHANE	22	36			7.6	8.2	
DIETHYL ETHER			20				
TETRAHYDROFURAN					2.8	<1	2.1
TRANS-1,2-DICHLOROETHENE	76	117			30	28	51
1,1-DICHLOROETHANE					<1	3.2	3.7
CIS-1,2-DICHLOROETHENE	2.8	137			147	164	32
CHLOROFORM	42	520			87	87	100
1,1,1-TRICHLOROETHANE							7.5
CARBON TETRACHLORIDE	232	132			27	20	17
BENZENE	19	58			22	16	9.8
1,2-DICHLOROETHANE	325	583			82	74	28
TRICHLOROETHYLENE							3.2
1,2-DICHLOROPROPANE						<3	
DIBROMOMETHANE							
TRANS-1,3-DICHLOROPROPENE		2.5					<0.7
TOLUENE							
CIS-1,3-DICHLOROPROPENE							<2
1,1,2-TRICHLOROETHANE	<2.5	105					
TETRACHLOROETHENE							
DIBROMOCHLOROMETHANE							
DIBROMOETHANE		134			2.9	<2	
CHLOROBENZENE		2					
ETHYL BENZENE							
m+p-XYLENE		10					
o-XYLENE							
STYRENE							
BROMOFORM							
CUMENE							
PROPYLBENZENE							
3+4-ETHYLTOLUENE							
2-ETHYLTOLUENE							
TRIMETHYL BENZENE							
1,3-DICHLOROBENZENE							
1,4-DICHLOROBENZENE							
1,3-DIETHYLBENZENE							
1,2-DICHLOROBENZENE							
1,4-DIETHYLBENZENE							
1,2-DIETHYLBENZENE							
SURROGATE							
BROMOCHLOROMETHANE	85	74	89	75	88	85	86

* SURROGATE was spiked at 100 ug/L

APPENDIX A

COMPOUNDS TENTATIVELY IDENTIFIED BY GC-MS
AND COMMENTS

SAMPLE	**88GL126P**	**UNKNOWNS**	
		Dioxane approx.	330

SAMPLE	**88GL16M12**	**UNKNOWNS**	
		1,1,2-trichloro-1,2,2-trifluoroethane	72

SAMPLE	**88GL16M16**	**UNKNOWNS**	
		1,1,2-trichloro-1,2,2-trifluoroethane	23
		dibromoethane	
		dibromoethene	

SAMPLE	**88GL21M02**	**UNKNOWNS**	
		Dioxane approx.	330

SAMPLE	**88GL21M03**	**UNKNOWNS**	
		Dioxane approx.	430

SAMPLE	**88GL37M01**	**UNKNOWNS**	
		1,1,2-trichloro-1,2,2-trifluoroethane	20

SAMPLE	**88GL37M06**	**UNKNOWNS**	
		1,1,2-trichloro-1,2,2-trifluoroethane	14
		Dichlorotrifluorethane	
		Dichlorodifluoromethane	

COMMENTS
SAMPLE ANALYSED AT 2X DILUTION

SAMPLE	**88GL37M07**	**UNKNOWNS**	
		1,1,2-trichloro-1,2,2-trifluoroethane	<5
		Diethyl disulfide	
		Diisopropyl ether	

COMMENTS
SAMPLE ANALYSED AT 2X DILUTION

SAMPLE 88GL37M09

UNKNOWNS
Dichlorotrifluoroethane
Dichlorodifluoromethane
Diethyl disulfide
Methyl ethyl sulfide
Dimethyl sulfide
1,2-Dichloroethane

COMMENTS
SAMPLE ANALYSED AT 5X DILUTION

SAMPLE 88GL47M01

UNKNOWNS
Dioxane approx. 500
Unknown, mwt 132

SAMPLE 88GL47M04

UNKNOWNS
Dioxane approx. 2000

SAMPLE 88GL54M06

UNKNOWNS
Dichlorotrifluoroethane

SAMPLE 88GL67M01

UNKNOWNS
1,1,2-trichloro-1,2,2-trifluoroethane 36

SAMPLE 88GL67M06

UNKNOWNS
1,1,2-trichloro-1,2,2-trifluoroethane 40

SAMPLE 88GL67M07

UNKNOWNS
1,1,2-trichloro-1,2,2-trifluoroethane 35

SAMPLE 88GL67M08

UNKNOWNS
1,1,2-trichloro-1,2,2-trifluoroethane 50

COMMENTS
SAMPLE ANALYSED AT 5X DILUTION

SAMPLE 88GL67M09

UNKNOWNS

1,1,2-trichloro-1,2,2-trifluoroethane 140
dichlorotrifluoroethane
Dichlorodifluoromethane

COMMENTS
SAMPLE ANALYSED AT 5X DILUTION

SAMPLE 88GL67M11

UNKNOWNS

Tetrachloroethane 15
1,1,2-trichloro-1,2,2-trifluoroethane 2725
Dichlorotrifluoroethane
Chlorotrifluoroethene
Dichlorodifluoromethane
Diisopropyl ether
Pentane
Methoxybenzene
Ethylcyclobutane

COMMENTS
SAMPLE ANALYSED AT 5X DILUTION

SAMPLE 88GL92M15

UNKNOWNS

Tetrachloroethane 6.9
1,1,2-trichloro-1,2,2-trifluoroethane 120
Dichlorotrifluoroethane
Chloropropene

SAMPLE 88GL92M17

UNKNOWNS

Tetrachloroethane ≈5
1,1,2-trichloro-1,2,2-trifluoroethane 115
Dichlorotrifluoroethane

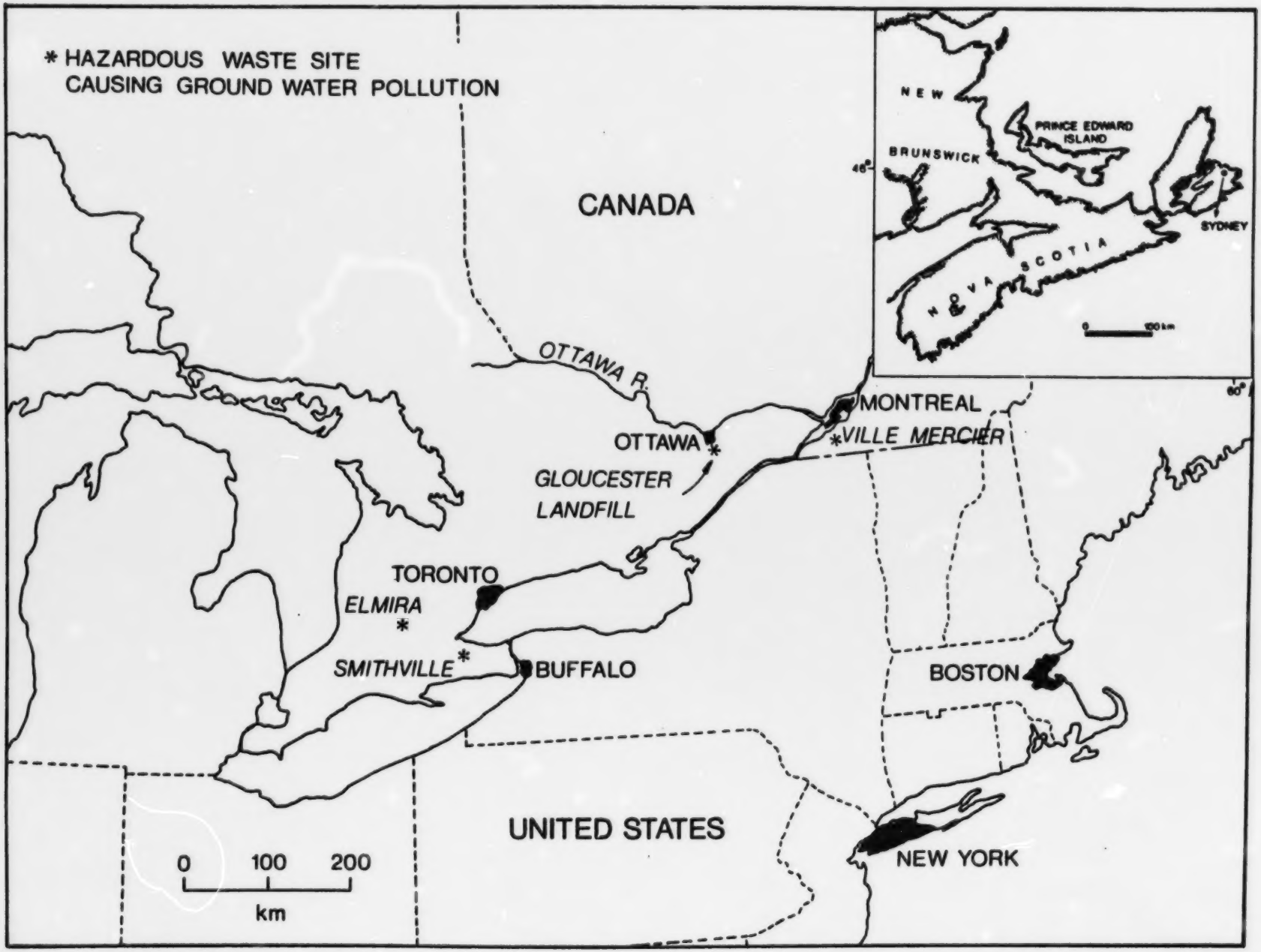
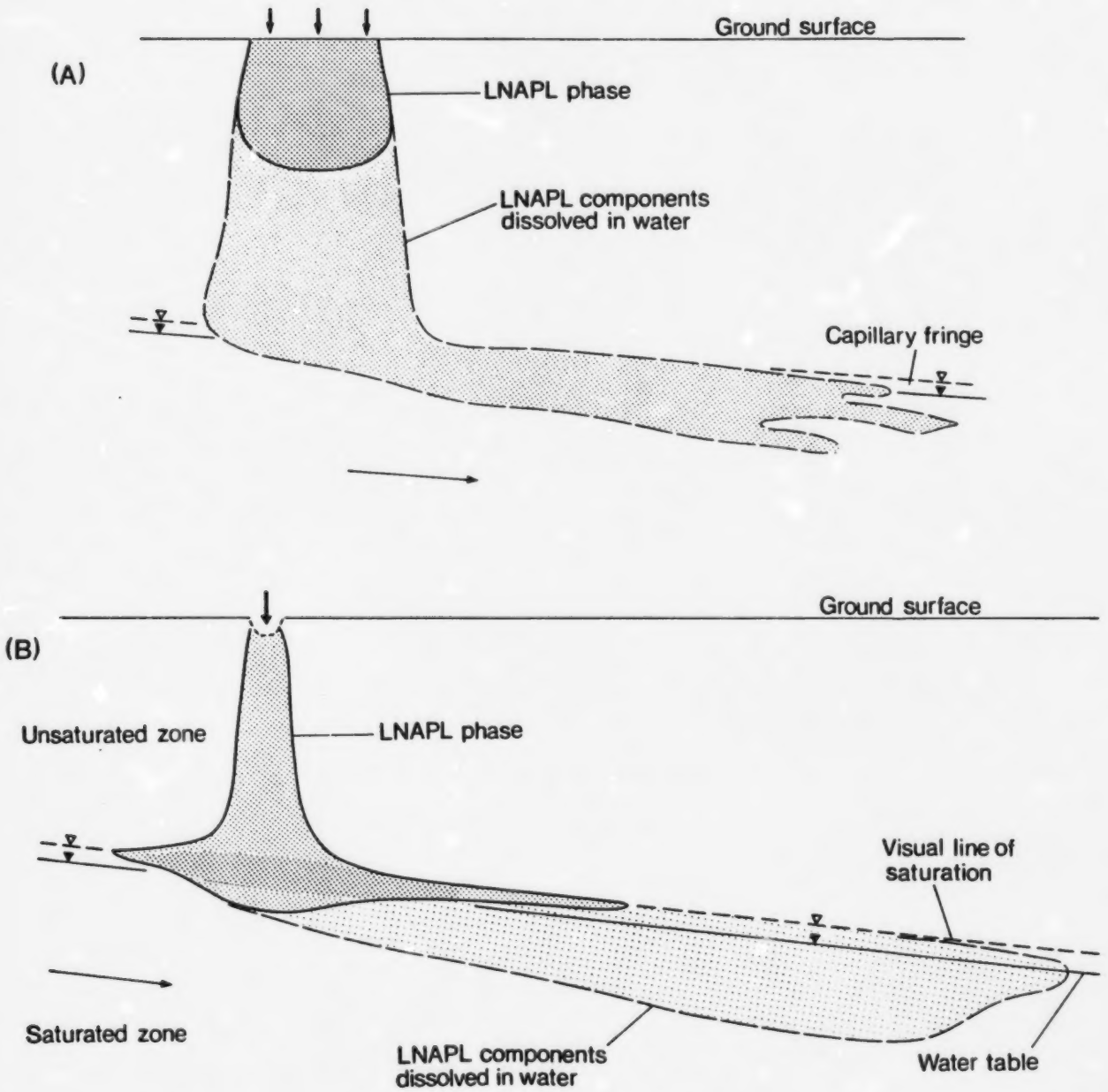

* HAZARDOUS WASTE SITE
CAUSING GROUND WATER POLLUTION
CANADA
UNITED STATES
OTTAWA R.
OTTAWA
GLOUCESTER
LANDFILL
MONTREAL
*VILLE MERCIER
TORONTO
ELMIRA
SMITHVILLE
BUFFALO
BOSTON
NEW YORK
0 100 200
km
NEW
BRUNSWICK
PRINCE EDWARD
ISLAND
SYDNEY
N & VA SCOTIA

Figure 1

Figure 2


(A)
Ground surface
LNAPL phase
LNAPL components dissolved in water
Capillary fringe
(B)
Ground surface
Unsaturated zone
LNAPL phase
Visual line of saturation
Saturated zone
LNAPL components dissolved in water
Water table

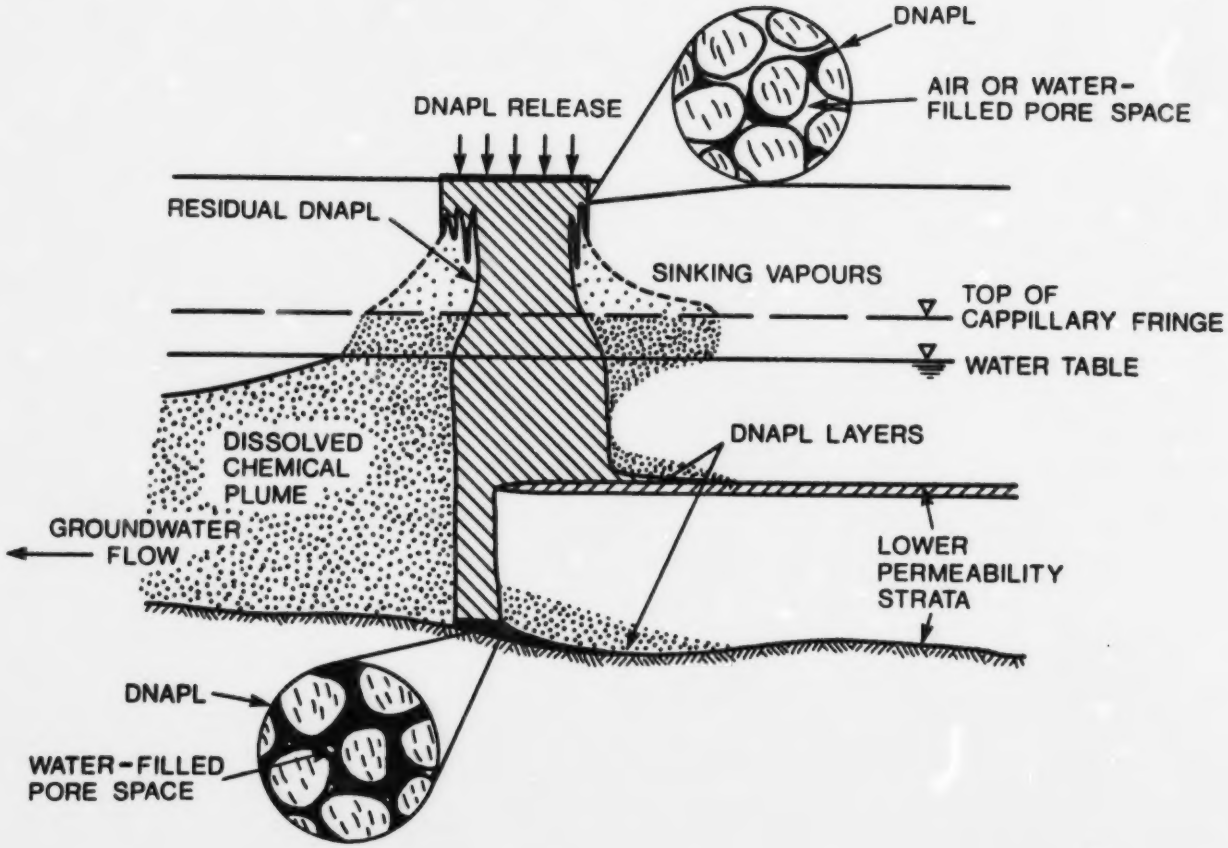
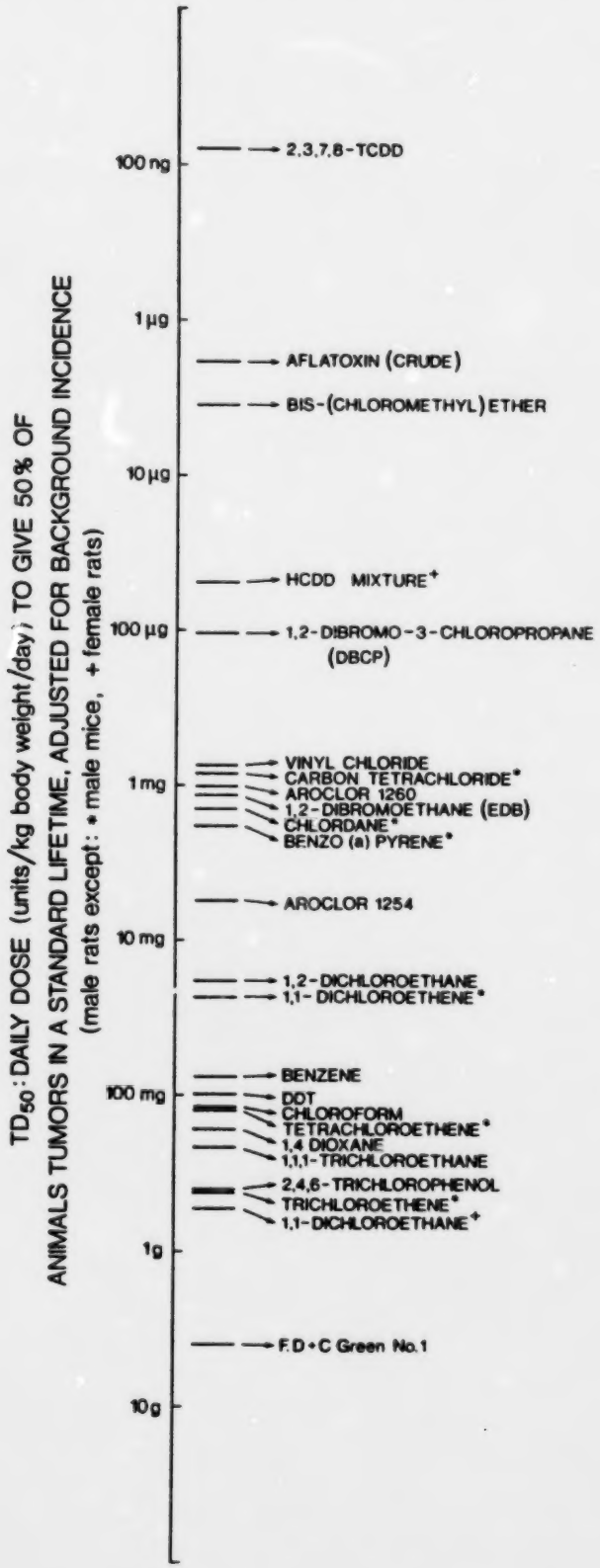

DNAPL
AIR OR WATER-
FILLED PORE SPACE
DNAPL RELEASE
RESIDUAL DNAPL
SINKING VAPOURS
TOP OF
CAPPILLARY FRINGE
WATER TABLE
DNAPL LAYERS
DISSOLVED
CHEMICAL
PLUME
GROUNDWATER
FLOW
LOWER
PERMEABILITY
STRATA
DNAPL
WATER-FILLED
PORE SPACE

Figure 3

Figure 4


TD_{50}: DAILY DOSE (units/kg body weight/day) TO GIVE 50% OF ANIMALS TUMORS IN A STANDARD LIFETIME, ADJUSTED FOR BACKGROUND INCIDENCE
(male rats except: * male mice, + female rats)
100 ng
1 µg
10 µg
100 µg
1 mg
10 mg
100 mg
1 g
10 g
2,3,7,8-TCDD
AFLATOXIN (CRUDE)
BIS-(CHLOROMETHYL) ETHER
HCDD MIXTURE+
1,2-DIBROMO-3-CHLOROPROPANE (DBCP)
VINYL CHLORIDE
CARBON TETRACHLORIDE*
AROCLOR 1260
1,2-DIBROMOETHANE (EDB)
CHLORDANE*
BENZO (a) PYRENE*
AROCLOR 1254
1,2-DICHLOROETHANE
1,1-DICHLOROETHENE*
BENZENE
DDT
CHLOROFORM
TETRACHLOROETHENE*
1,4 DIOXANE
1,1,1-TRICHLOROETHANE
2,4,6-TRICHLOROPHENOL
TRICHLOROETHENE*
1,1-DICHLOROETHANE+
F.D+C Green No.1

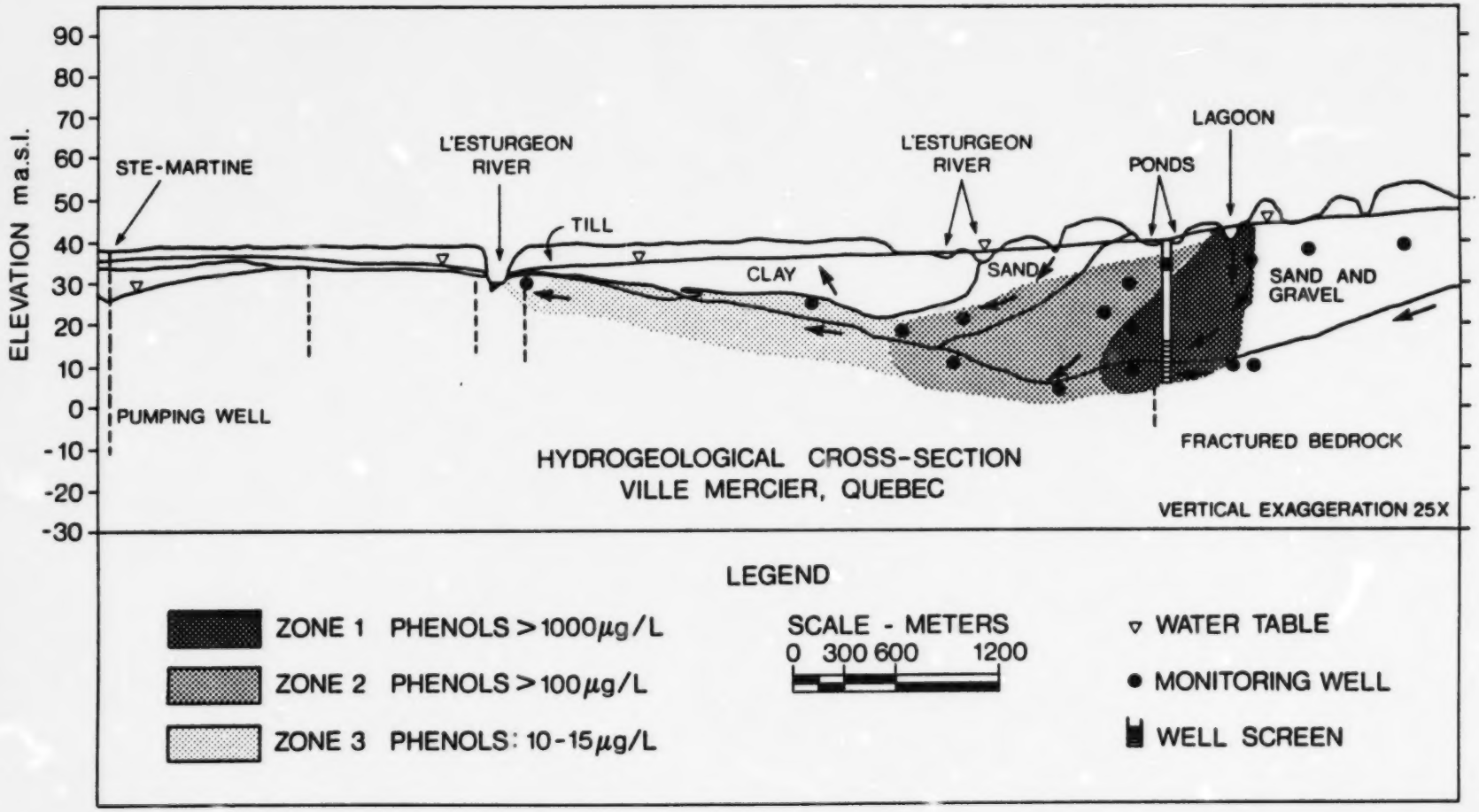
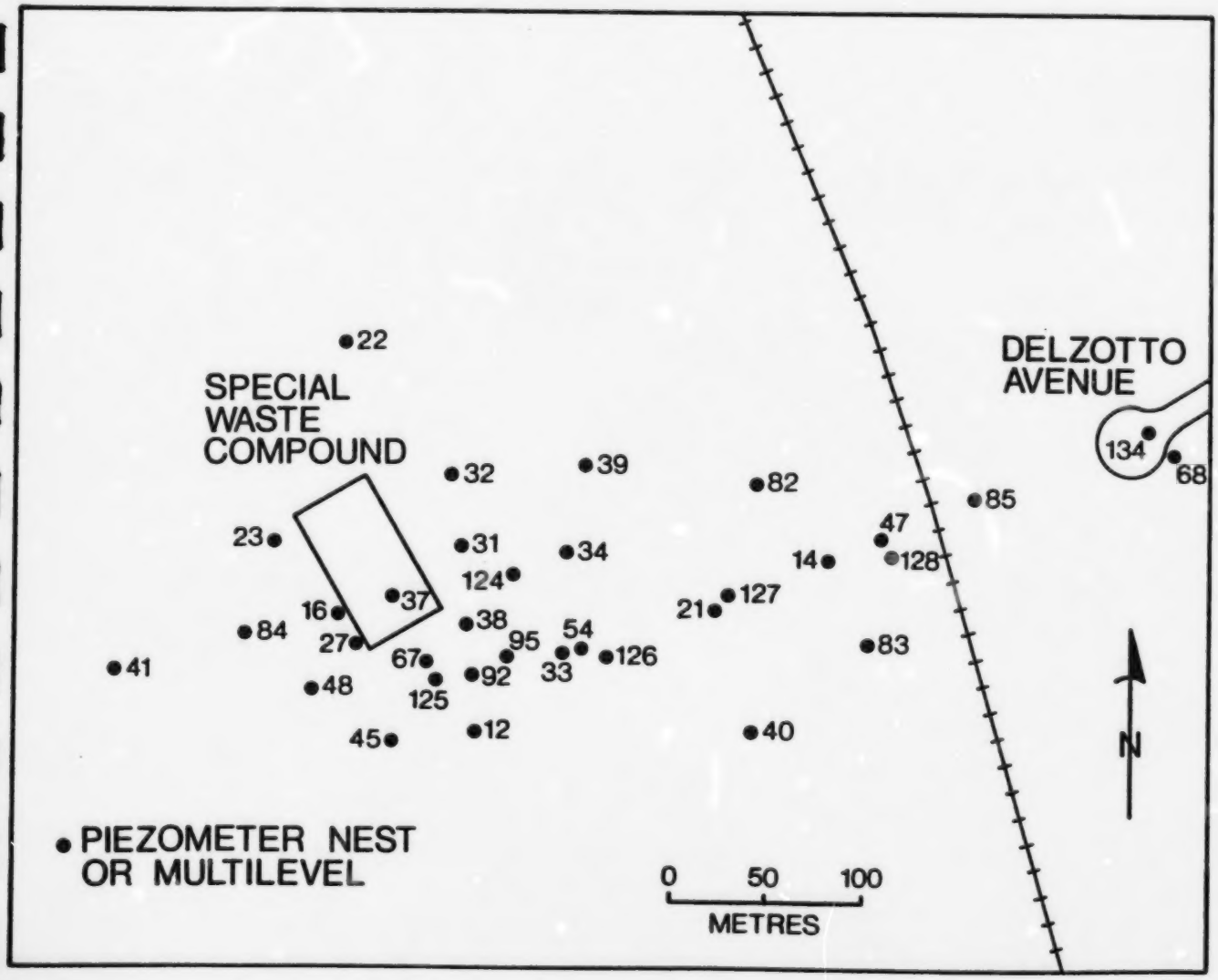
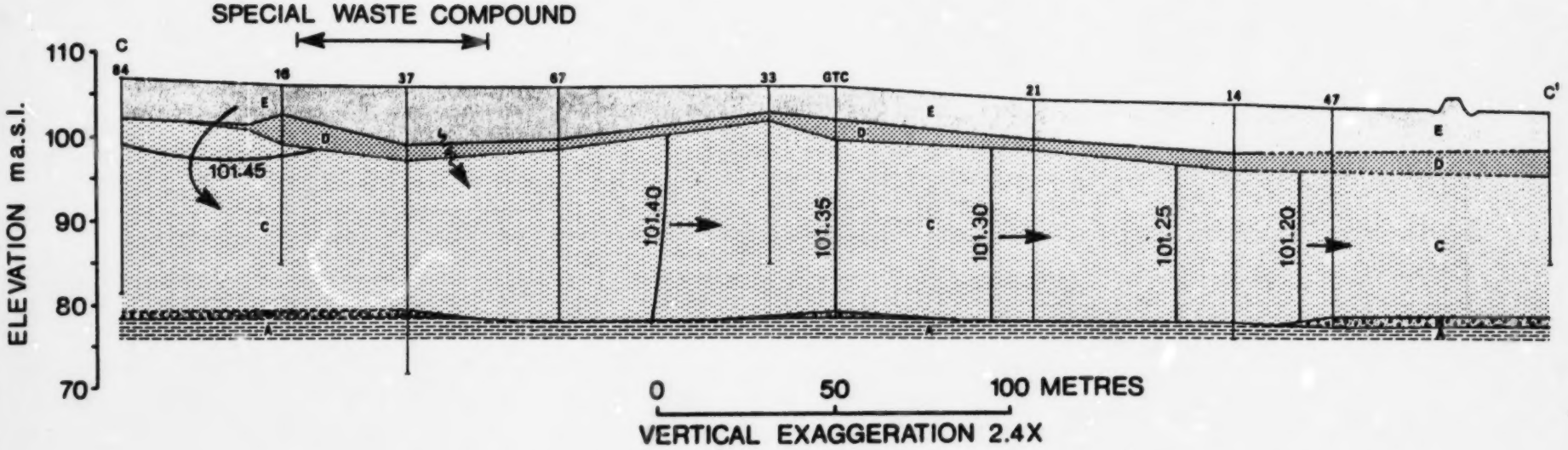

ELEVATION m.a.s.l.
90
80
70
60
50
40
30
20
10
0
-10
-20
-30
STE-MARTINE
L'ESTURGEON RIVER
L'ESTURGEON RIVER
TILL
CLAY
SAND
PONDS
LAGOON
SAND AND GRAVEL
PUMPING WELL
FRACTURED BEDROCK
HYDROGEOLOGICAL CROSS-SECTION
VILLE MERCIER, QUEBEC
VERTICAL EXAGGERATION 25X
LEGEND
ZONE 1 PHENOLS > 1000µg/L
ZONE 2 PHENOLS > 100µg/L
ZONE 3 PHENOLS: 10-15µg/L
SCALE - METERS
0 300 600 1200
WATER TABLE
MONITORING WELL
WELL SCREEN

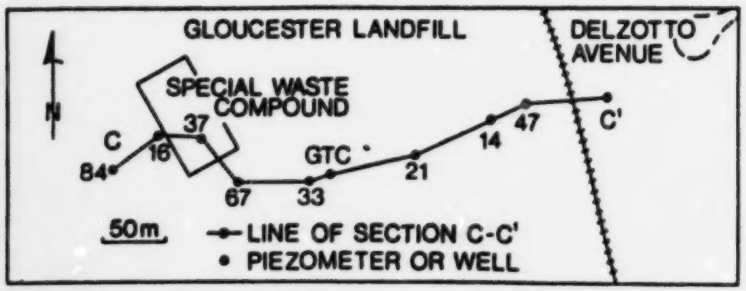
Figure 5

Figure 6

STUDY AREA







LEGEND

STRATIGRAPHIC UNIT

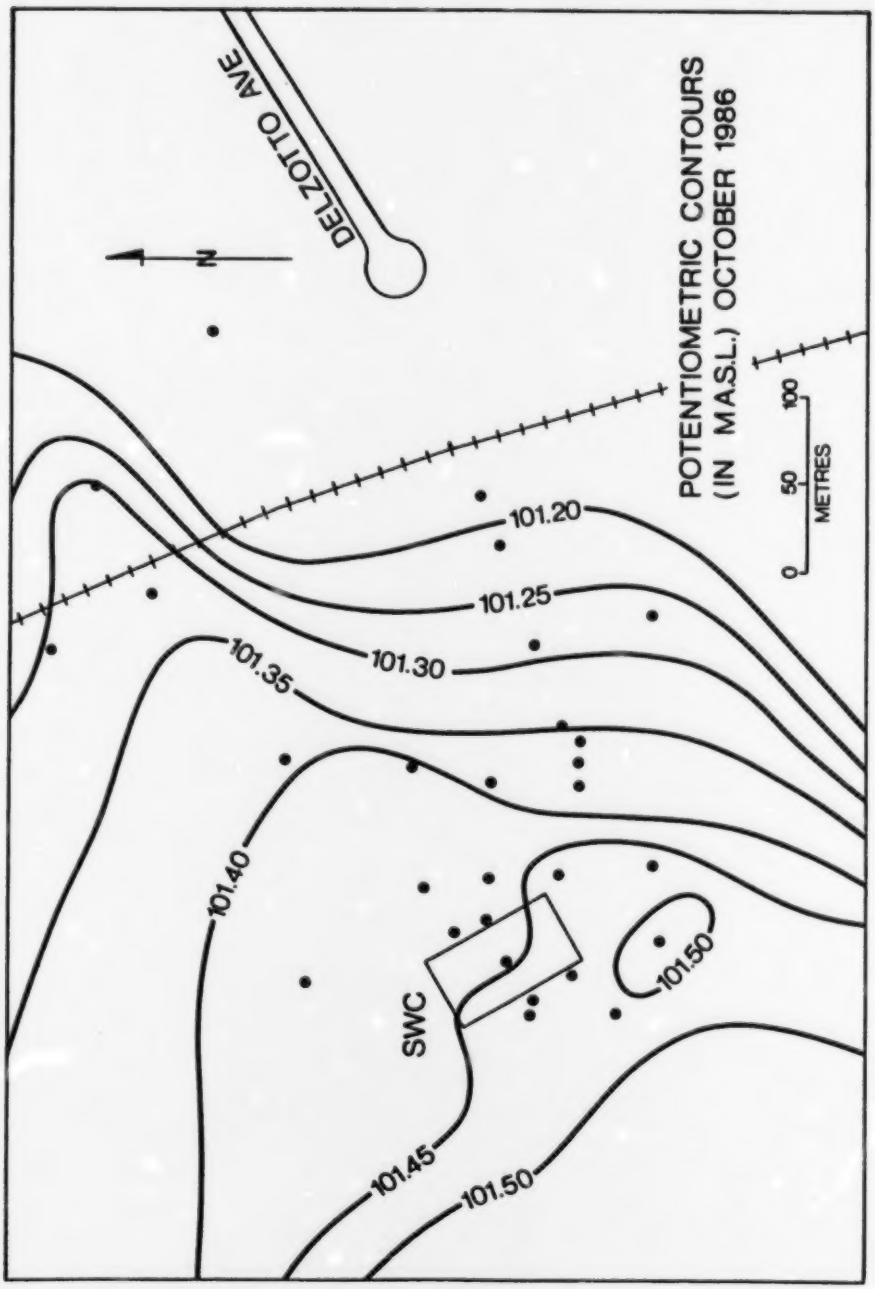
- A LIMESTONE
- B TILL
- C OUTWASH
- D SILT
- E SANDS

GROUND WATER FLOW DIRECTION

101.40 EQUIPOTENTIAL CONTOUR OF HYDRAULIC HEADS MEASURED OCTOBER 1986

Figure 7

Figure 8


DELZOTTO AVE
N
POTENTIOMETRIC CONTOURS
(IN M.A.S.L.) OCTOBER 1986
0
50
100
METRES
101.20
101.25
101.30
101.35
101.40
101.45
101.50
101.50
SWC

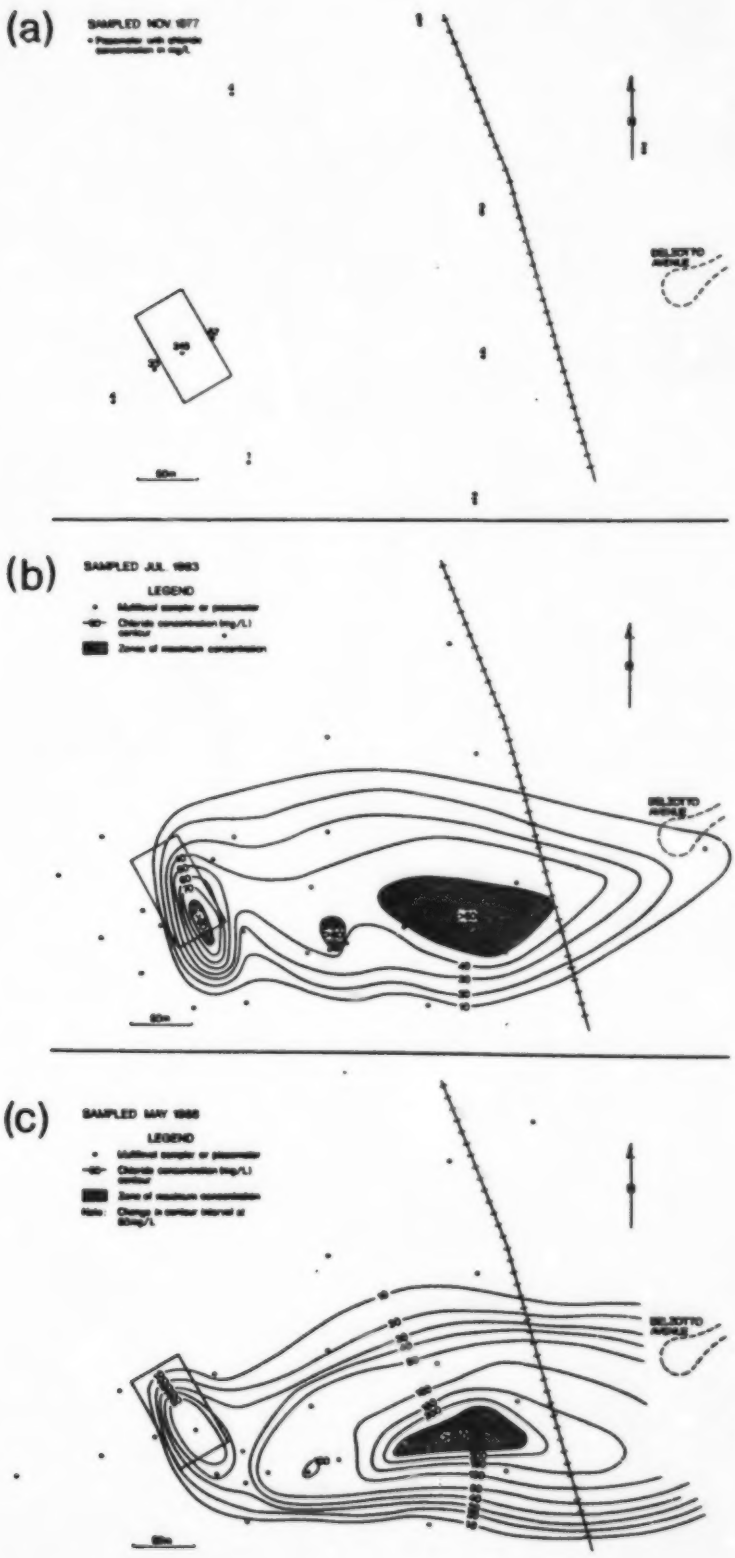
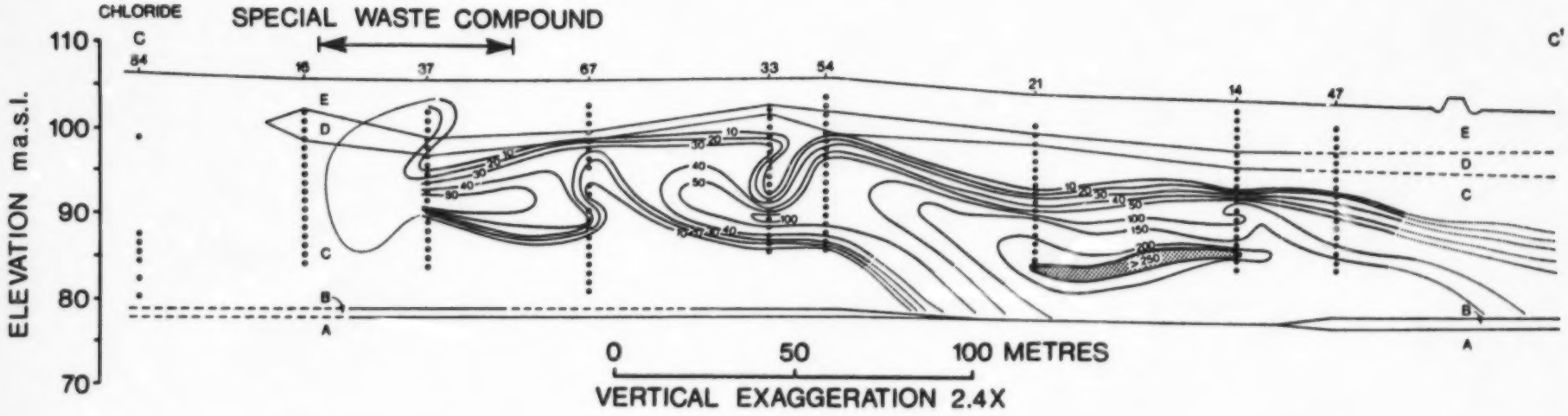
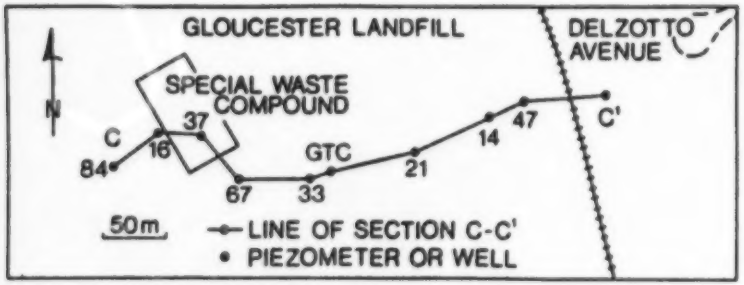


Figure 7. Distribution of maximum chloride concentrations at each test hole within the outwash aquifer. (a) Nov. 1977, (b) Jul. 1983, (c) May 1988.





STRATIGRAPHIC UNIT

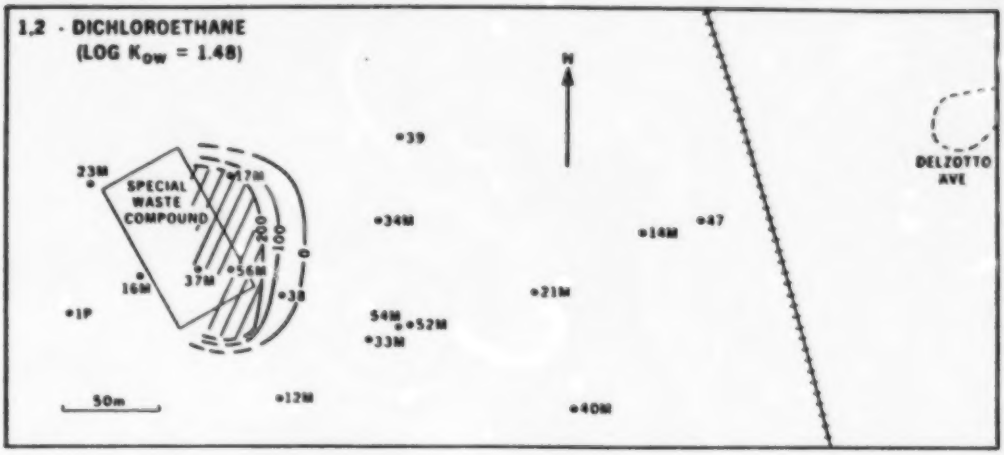
- A LIMESTONE
- B TILL
- C OUTWASH
- D SILT
- E SANDS

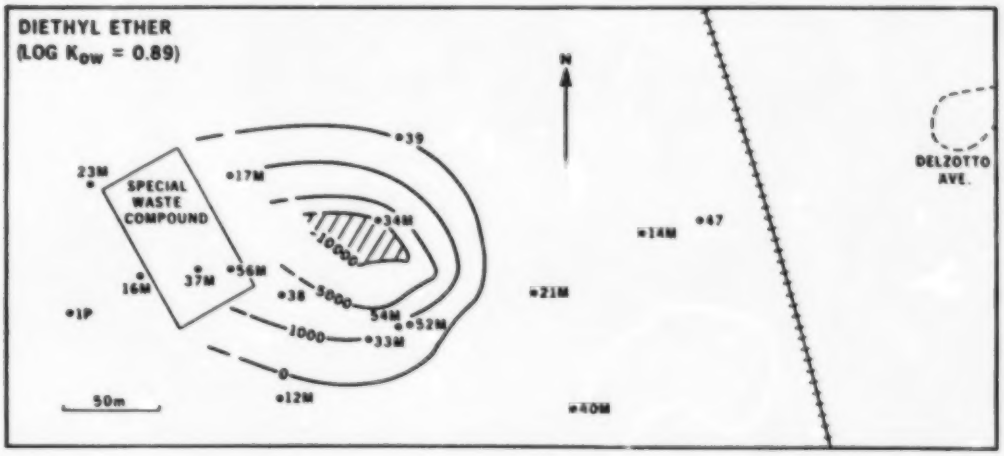
LEGEND

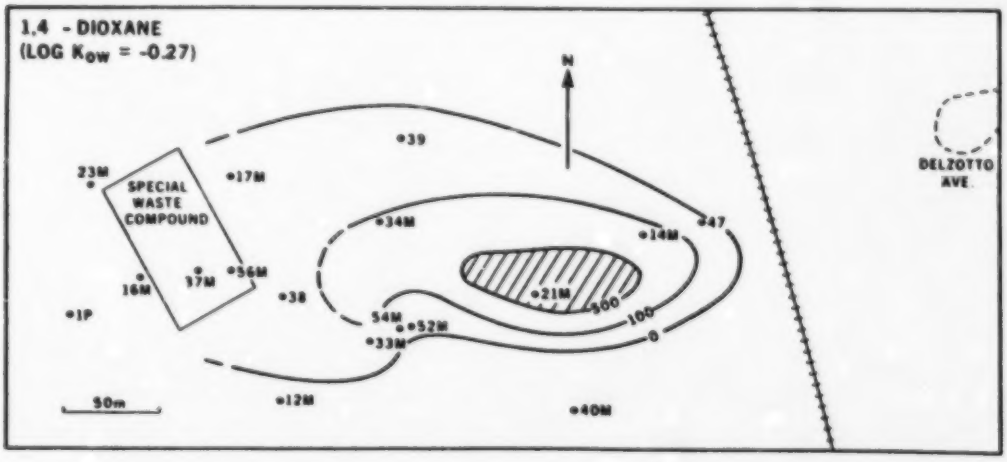
- • SAMPLE LOCATION
- −10− CONCENTRATION CONTOUR IN mgCl⁻/L (1988)
- NOTE CHANGE IN CONTOUR INTERVAL AT 50mg/L
- ZONE OF MAXIMUM CONCENTRATION
- 33 MULTILEVEL SAMPLER

Figure 10

Figure 11







LEGEND

- • MULTILEVEL SAMPLER OR PIEZOMETER
- —500— CONTOUR OF MAXIMUM CONCENTRATION (PARTS PER BILLION) ENCOUNTERED IN SECTION (1982)
- AREA OF MAXIMUM CONCENTRATION

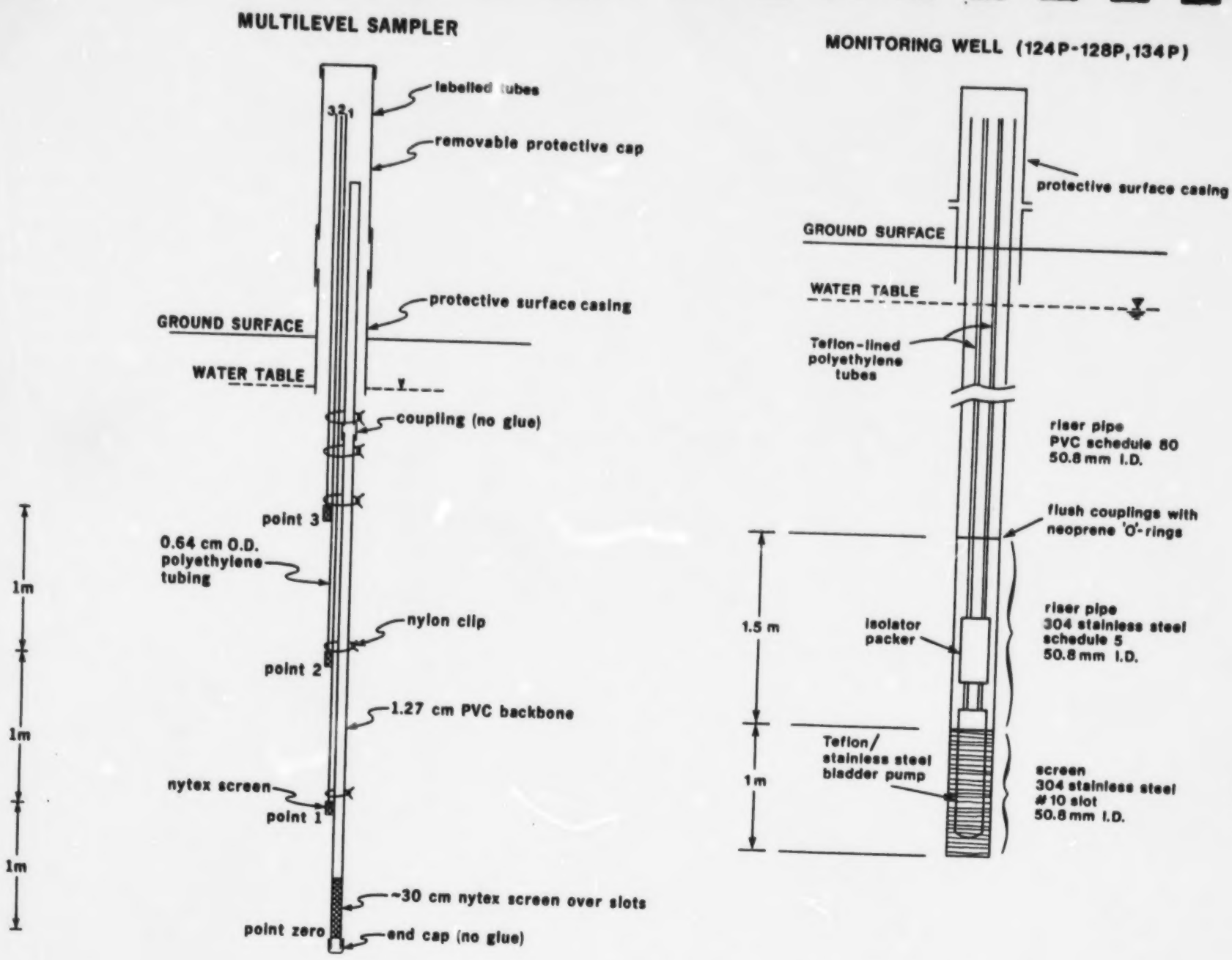

MULTILEVEL SAMPLER
labelled tubes
3 2 1
removable protective cap
protective surface casing
GROUND SURFACE
WATER TABLE
coupling (no glue)
point 3
0.64 cm O.D. polyethylene tubing
nylon clip
point 2
1.27 cm PVC backbone
nytex screen
point 1
~30 cm nytex screen over slots
point zero
end cap (no glue)
1m
1m
1m
MONITORING WELL (124P-128P, 134P)
protective surface casing
GROUND SURFACE
WATER TABLE
Teflon-lined polyethylene tubes
riser pipe PVC schedule 80 50.8 mm I.D.
flush couplings with neoprene 'O'-rings
1.5 m
isolator packer
riser pipe 304 stainless steel schedule 5 50.8 mm I.D.
Teflon/ stainless steel bladder pump
1m
screen 304 stainless steel #10 slot 50.8 mm I.D.

Figure 12

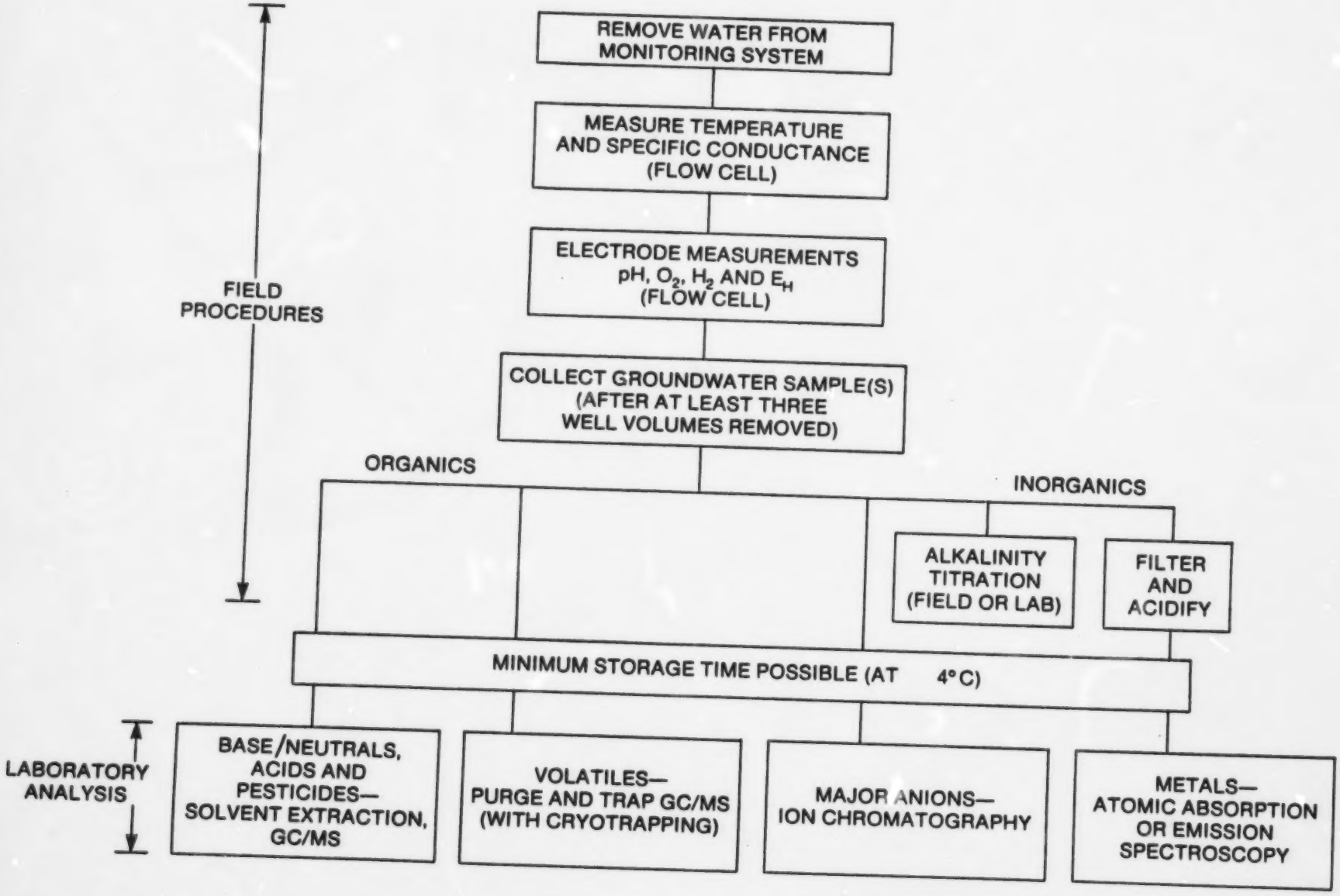
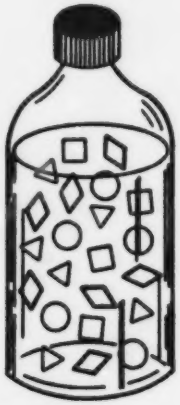

FIELD PROCEDURES
REMOVE WATER FROM MONITORING SYSTEM
MEASURE TEMPERATURE AND SPECIFIC CONDUCTANCE (FLOW CELL)
ELECTRODE MEASUREMENTS pH, O_2, H_2 AND E_H (FLOW CELL)
COLLECT GROUNDWATER SAMPLE(S) (AFTER AT LEAST THREE WELL VOLUMES REMOVED)
ORGANICS
INORGANICS
ALKALINITY TITRATION (FIELD OR LAB)
FILTER AND ACIDIFY
MINIMUM STORAGE TIME POSSIBLE (AT 4°C)
LABORATORY ANALYSIS
BASE/NEUTRALS, ACIDS AND PESTICIDES— SOLVENT EXTRACTION, GC/MS
VOLATILES— PURGE AND TRAP GC/MS (WITH CRYOTRAPPING)
MAJOR ANIONS— ION CHROMATOGRAPHY
METALS— ATOMIC ABSORPTION OR EMISSION SPECTROSCOPY

Figure 13

COMPONENTS OF A GC/MS SYSTEM

SAMPLE



GAS CHROMATOGRAPHY

MASS SPECTROMETRY

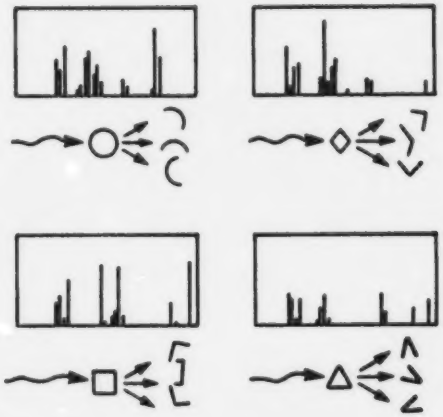
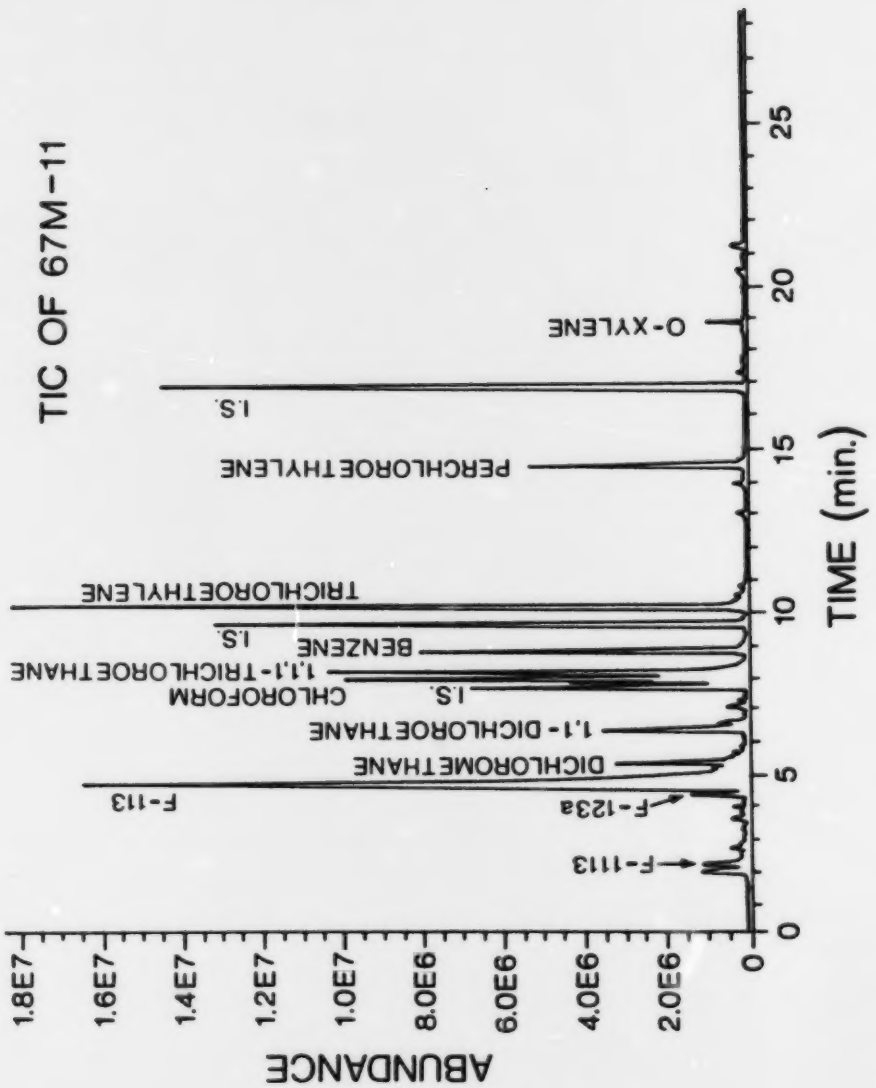
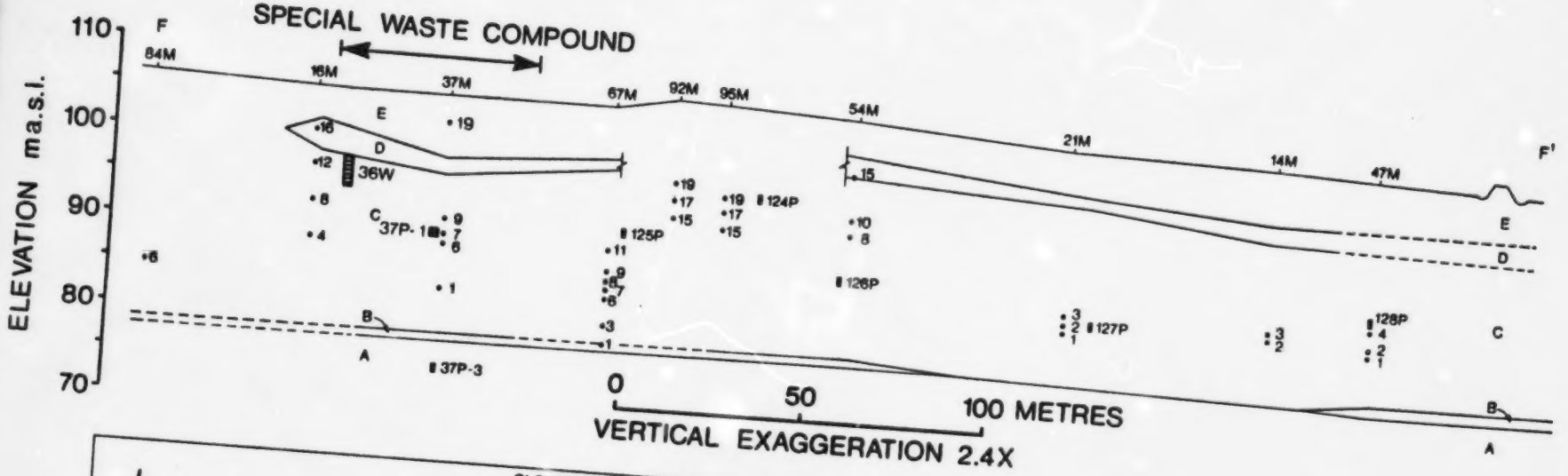
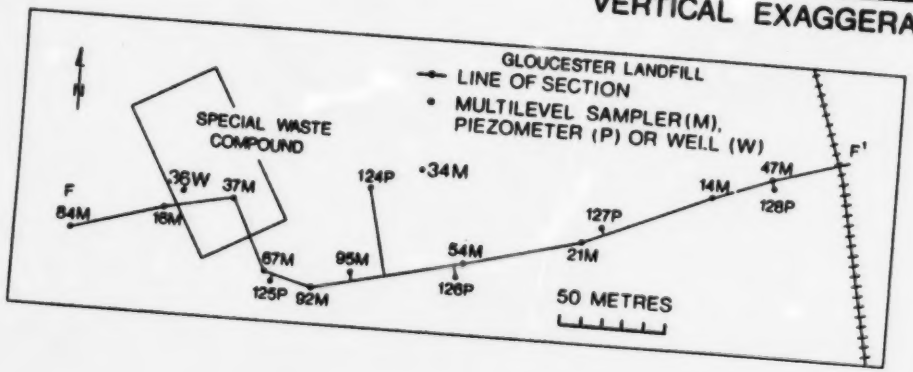


Figure 14

Figure 15


TIC OF 67M-11
ABUNDANCE
TIME (min.)
1.8E7
1.6E7
1.4E7
1.2E7
1.0E7
8.0E6
6.0E6
4.0E6
2.0E6
0
5
10
15
20
25
F-1113
F-123a
F-113
DICHLOROMETHANE
1,1-DICHLOROETHANE
I.S.
CHLOROFORM
1,1,1-TRICHLOROETHANE
BENZENE
I.S.
TRICHLOROETHYLENE
PERCHLOROETHYLENE
I.S.
O-XYLENE





LEGEND

STRATIGRAPHIC UNIT

- A LIMESTONE
- B TILL
- C OUTWASH
- D SILT
- E SANDS

- MULTILEVEL
- WELL SCREEN

Figure 16

Figure 17

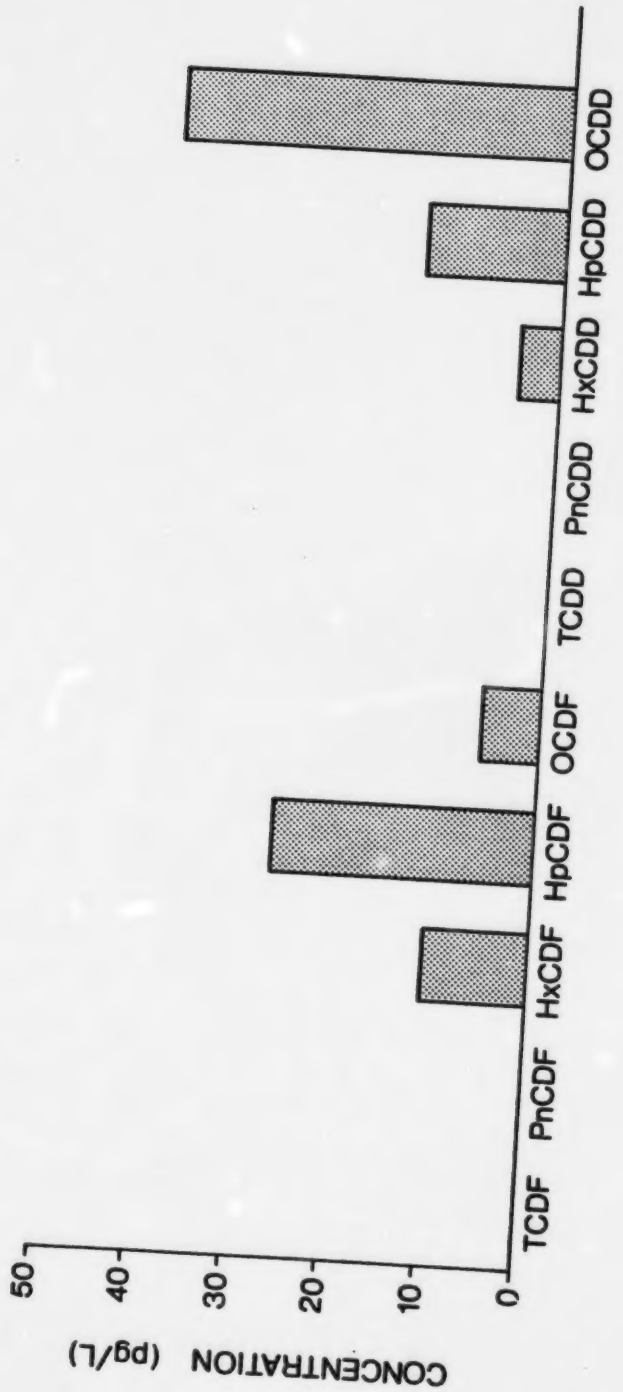

CONCENTRATION (pg/L)
0
10
20
30
40
50
TCDF
PnCDF
HxCDF
HpCDF
OCDF
TCDD
PnCDD
HxCDD
HpCDD
OCDD

Figure 18 a, b

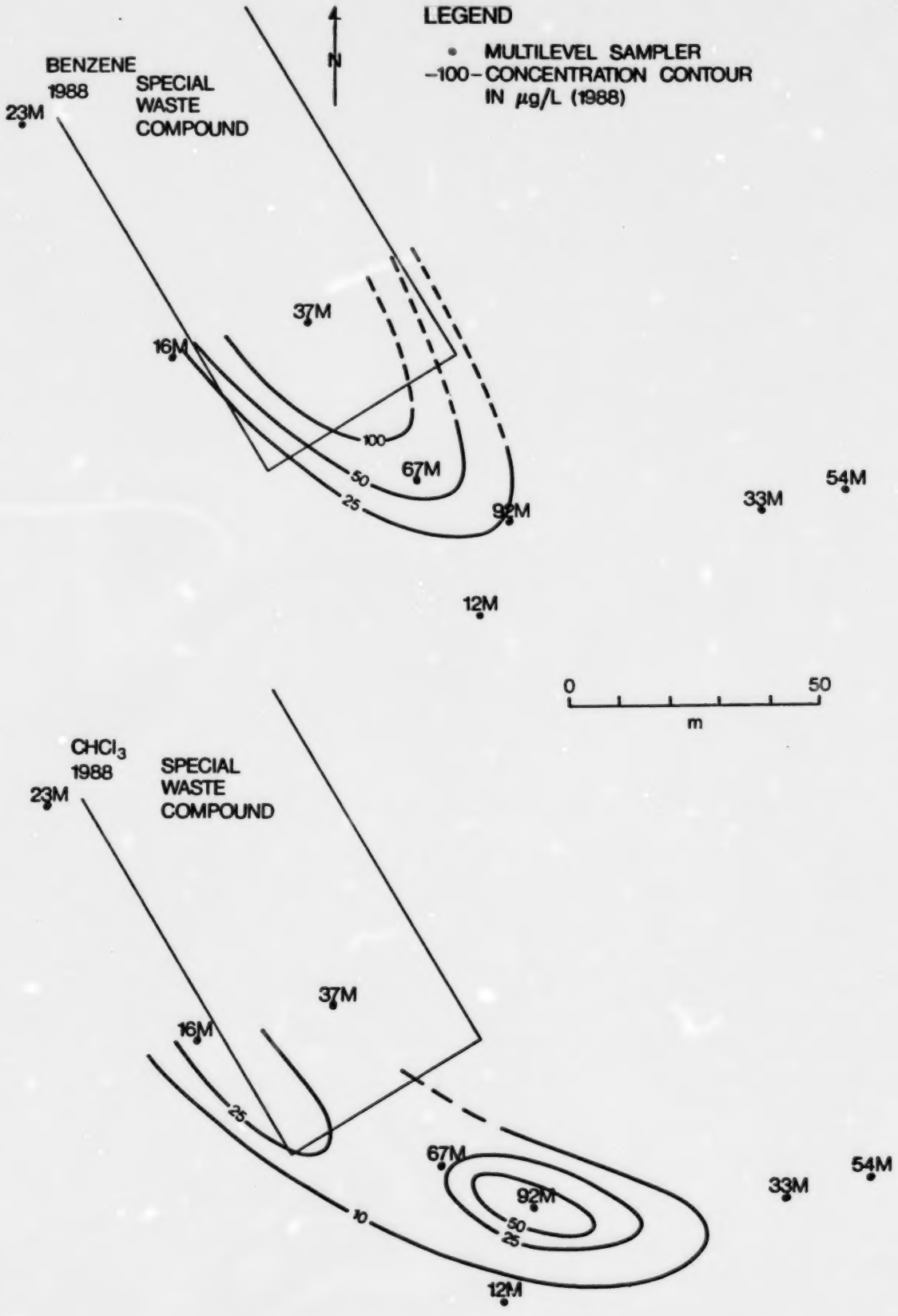

LEGEND
MULTILEVEL SAMPLER
-100- CONCENTRATION CONTOUR IN µg/L (1988)
N
BENZENE
1988
SPECIAL WASTE COMPOUND
23M
37M
16M
100
50
25
67M
92M
33M
54M
12M
0
50
m
CHCl3
1988
SPECIAL WASTE COMPOUND
23M
37M
16M
25
67M
92M
50
25
10
33M
54M
12M

Figure 18 c, d, e

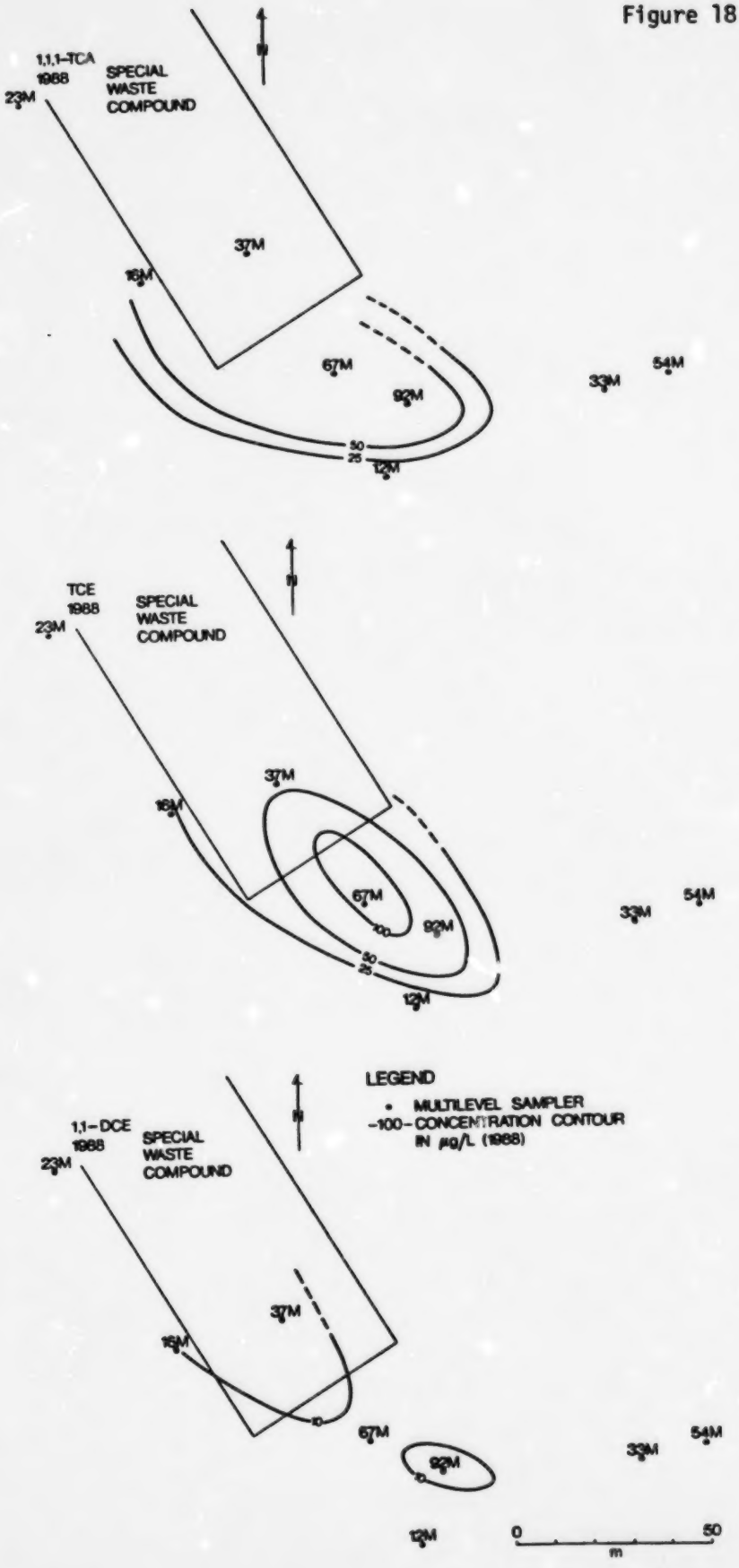


Figure 19 a, b.

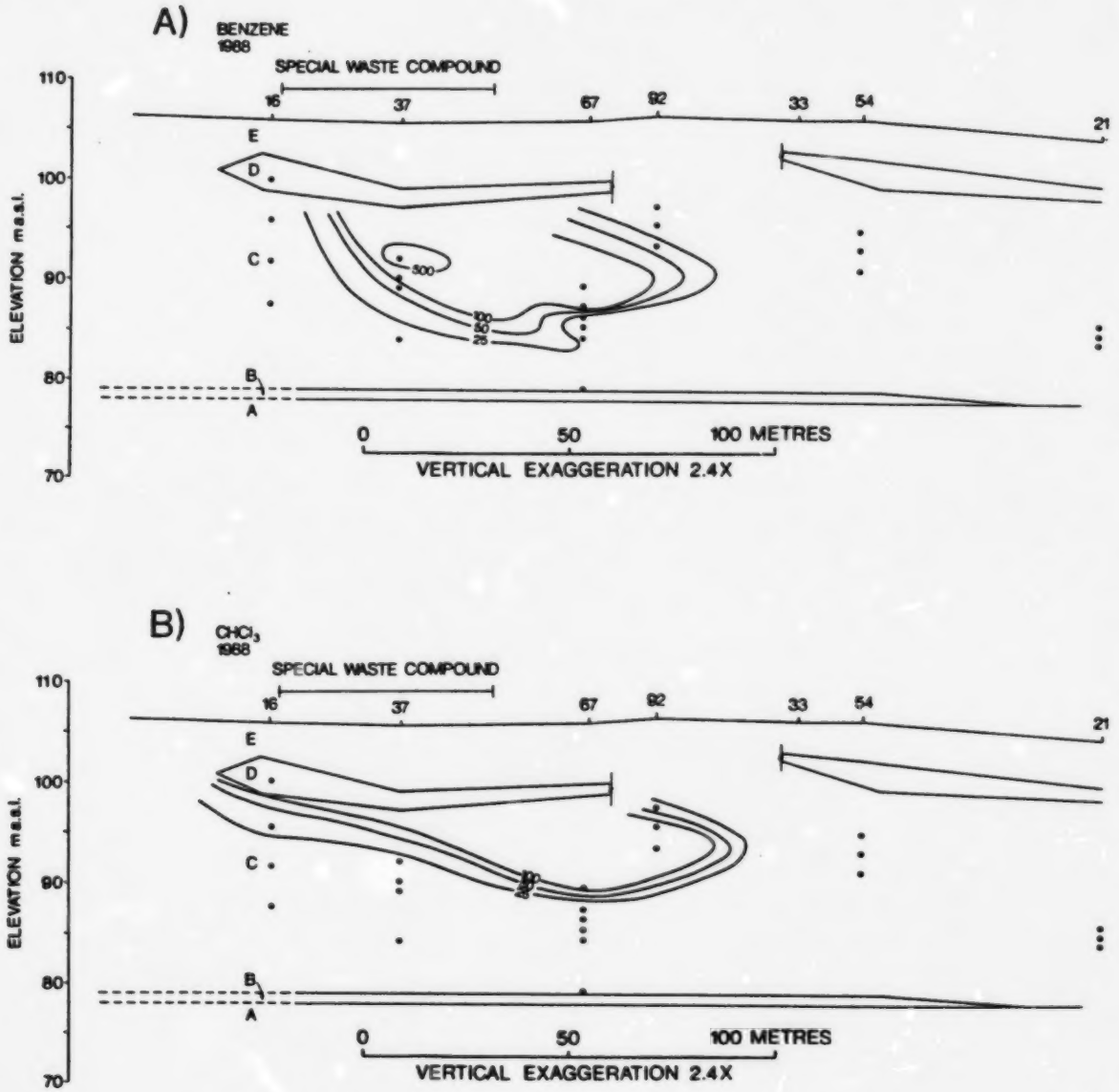
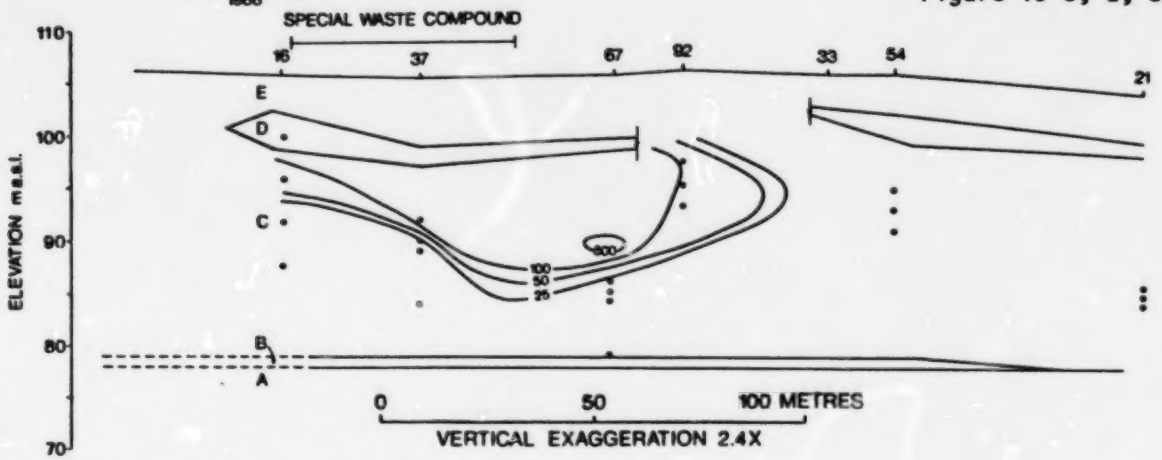
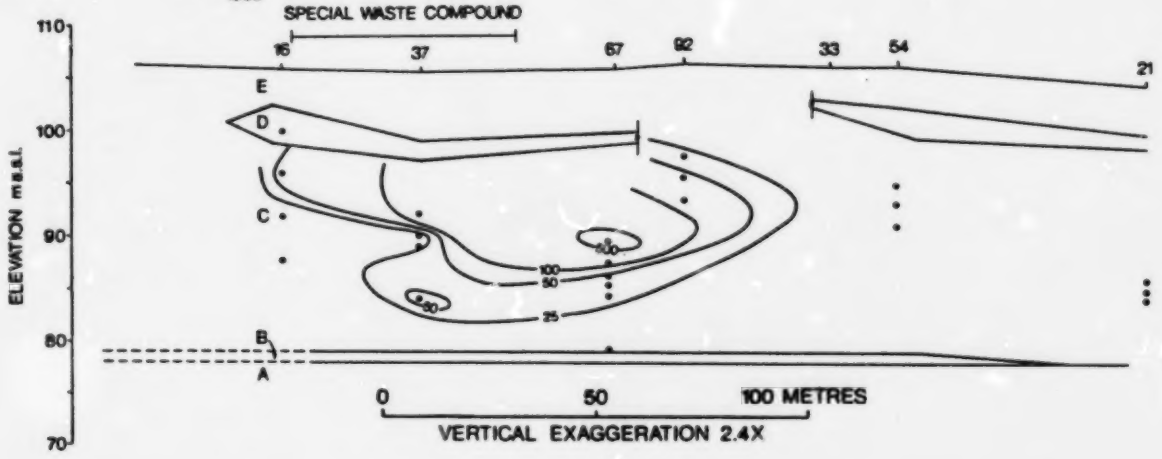

A)
BENZENE
1988
SPECIAL WASTE COMPOUND
16
37
67
92
33
54
21
E
D
C
B
A
500
100
50
25
110
100
90
80
70
ELEVATION m.a.s.l.
0
50
100 METRES
VERTICAL EXAGGERATION 2.4X
B)
CHCl3
1988
SPECIAL WASTE COMPOUND
16
37
67
92
33
54
21
E
D
C
B
A
110
100
90
80
70
ELEVATION m.a.s.l.
0
50
100 METRES
VERTICAL EXAGGERATION 2.4X

Figure 19 c, d, e

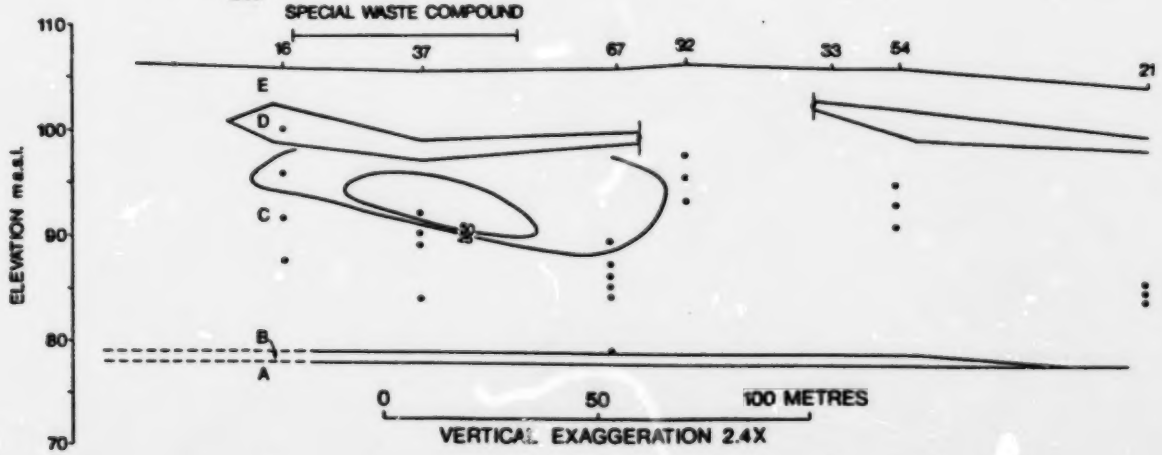
C) 1,1,1-TCA 1988



D) TCE 1988



E) 1,1-DCE 1988



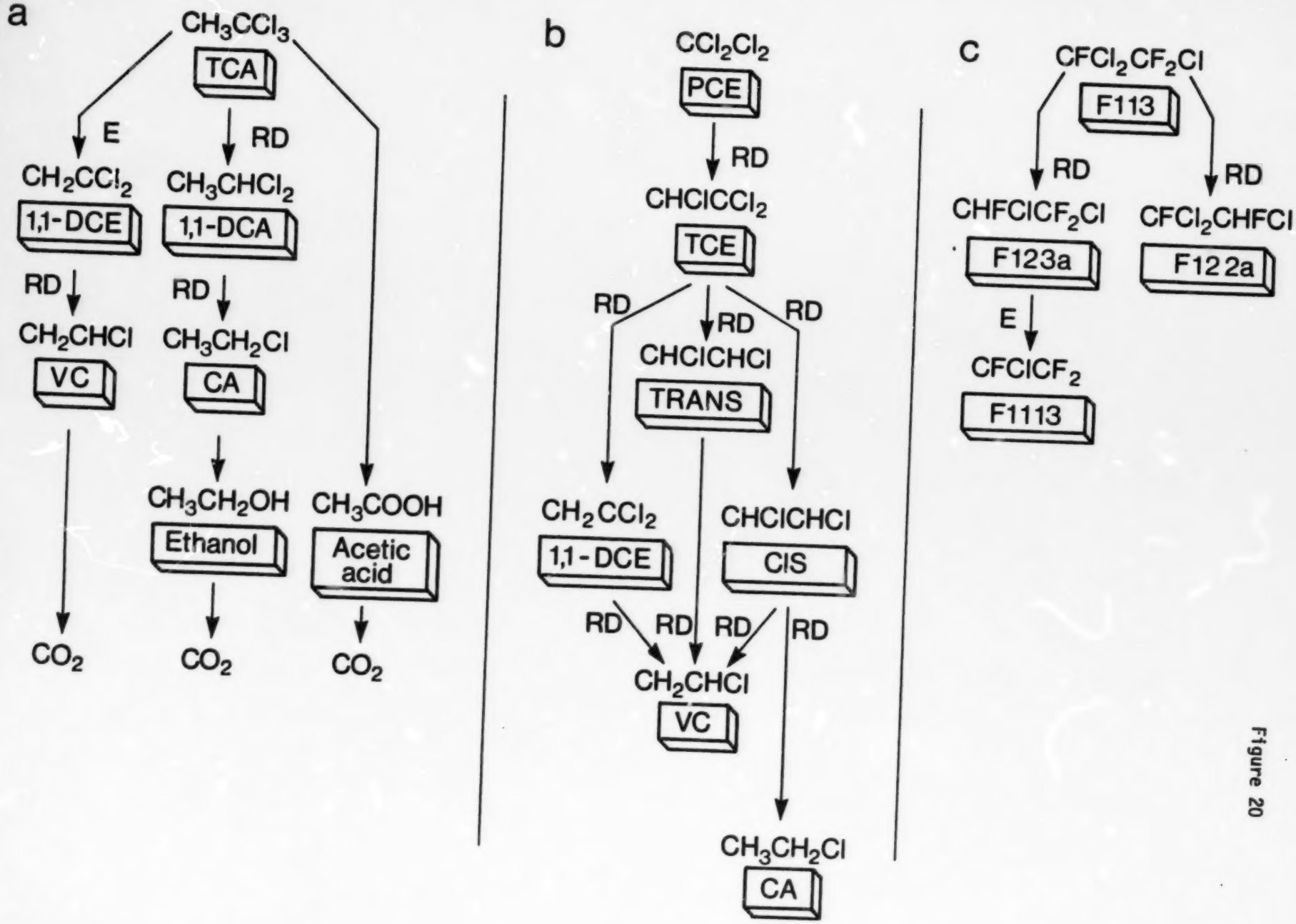

a
CH_3CCl_3
TCA
E
RD
CH_2CCl_2
1,1-DCE
CH_3CHCl_2
1,1-DCA
RD
RD
CH_2CHCl
VC
CH_3CH_2Cl
CA
CH_3CH_2OH
Ethanol
CH_3COOH
Acetic acid
CO_2
CO_2
CO_2
b
CCl_2Cl_2
PCE
RD
$CHClCCl_2$
TCE
RD
RD
RD
$CHClCHCl$
TRANS
CH_2CCl_2
1,1-DCE
$CHClCHCl$
CIS
RD
RD
RD
RD
CH_2CHCl
VC
CH_3CH_2Cl
CA
c
$CFCl_2CF_2Cl$
F113
RD
RD
$CHFClCF_2Cl$
F123a
$CFCl_2CHFCl$
F122a
E
$CFClCF_2$
F1113

Figure 20

Figure 21

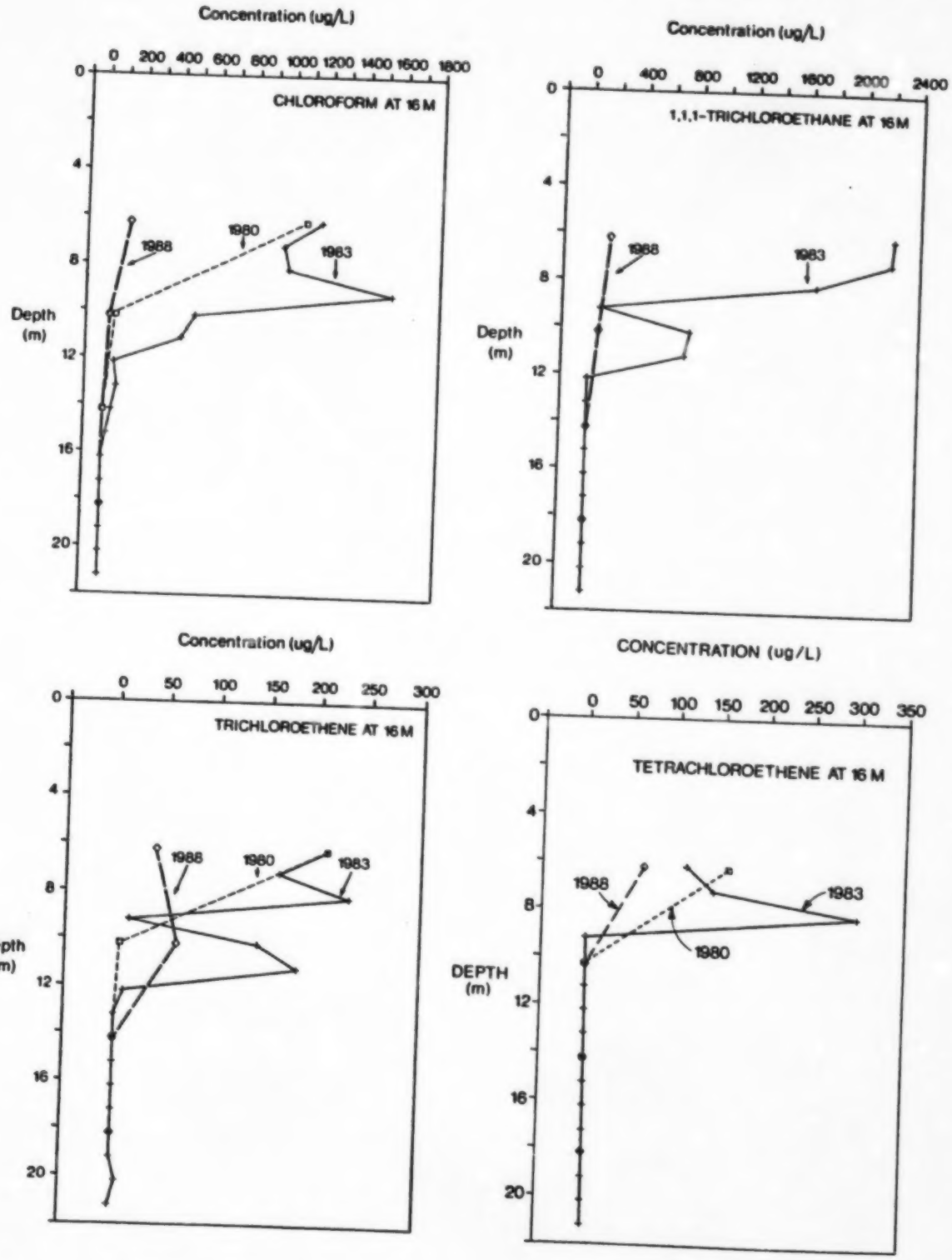
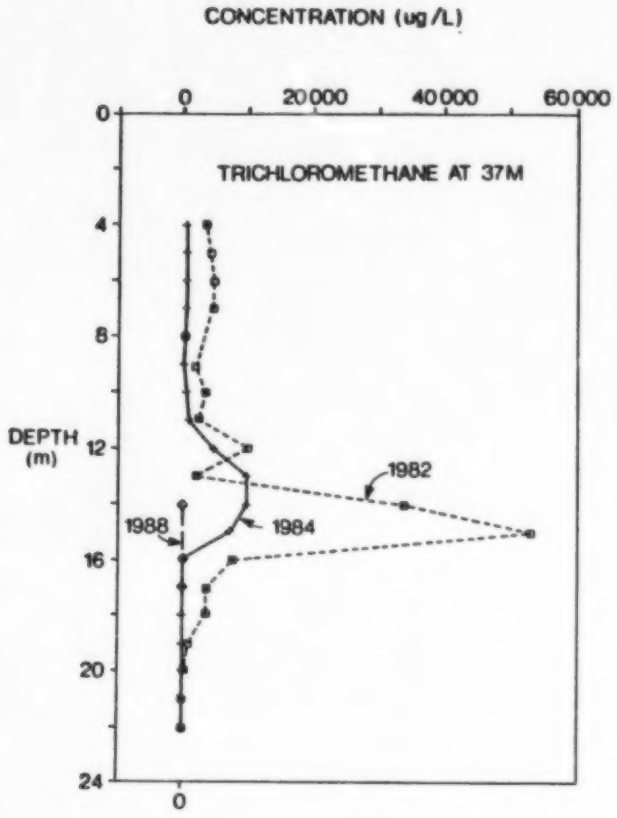
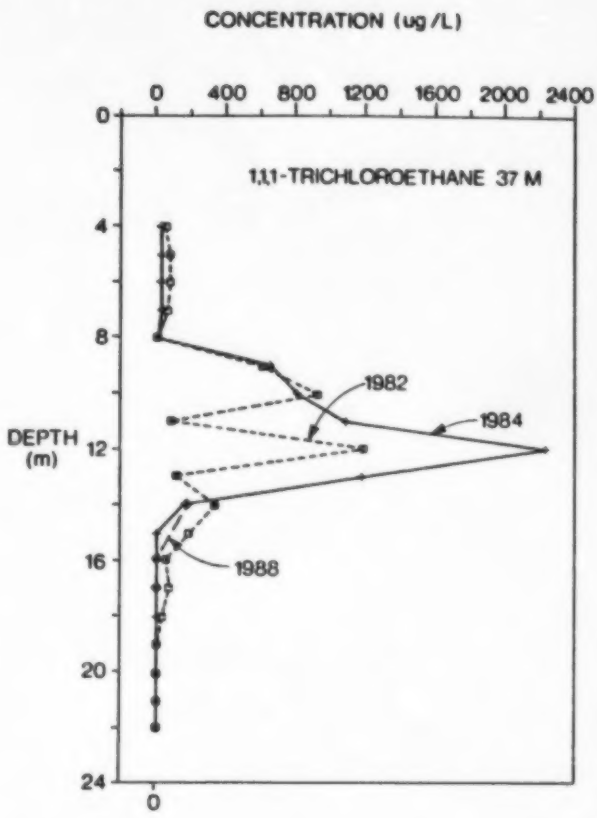
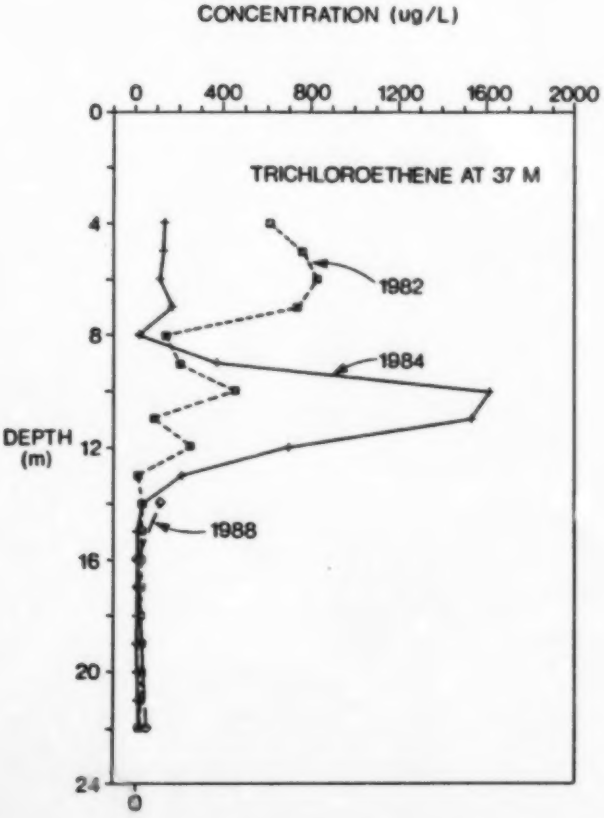

Concentration (ug/L)
0 200 400 600 800 1000 1200 1400 1600 1800
CHLOROFORM AT 16 M
1988
1980
1983
Depth (m)
0 4 8 12 16 20
Concentration (ug/L)
0 400 800 1200 1600 2000 2400
1,1,1-TRICHLOROETHANE AT 16 M
1988
1983
Depth (m)
0 4 8 12 16 20
Concentration (ug/L)
0 50 100 150 200 250 300
TRICHLOROETHENE AT 16 M
1988
1980
1983
Depth (m)
0 4 8 12 16 20
CONCENTRATION (ug/L)
0 50 100 150 200 250 300 350
TETRACHLOROETHENE AT 16 M
1988
1983
1980
DEPTH (m)
0 4 8 12 16 20

Figure 22


CONCENTRATION (ug/L)
0
20000
40000
60000
TRICHLOROMETHANE AT 37M
DEPTH (m)
0
4
8
12
16
20
24
1982
1984
1988
0


CONCENTRATION (ug/L)
0
400
800
1200
1600
2000
2400
1,1,1-TRICHLOROETHANE 37 M
DEPTH (m)
0
4
8
12
16
20
24
1982
1984
1988
0


CONCENTRATION (ug/L)
0
400
800
1200
1600
2000
TRICHLOROETHENE AT 37 M
DEPTH (m)
0
4
8
12
16
20
24
1982
1984
1988
0

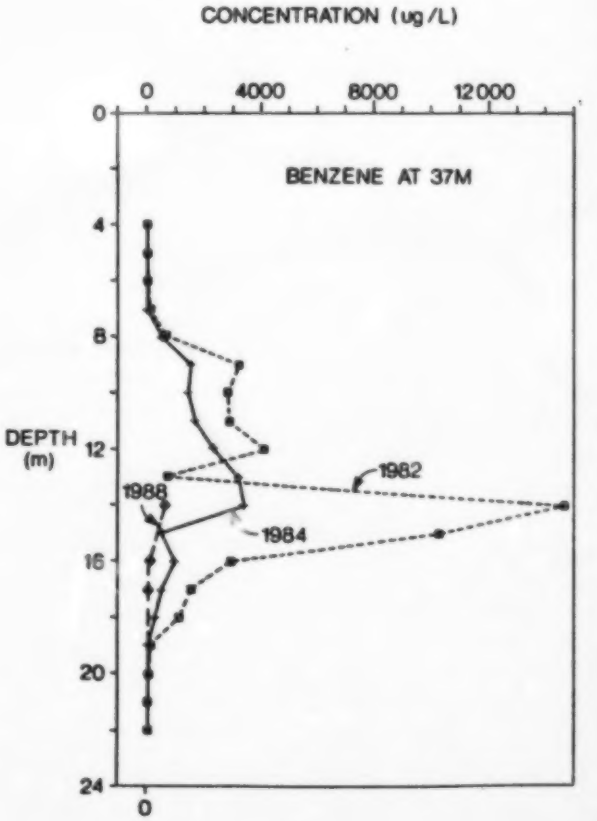
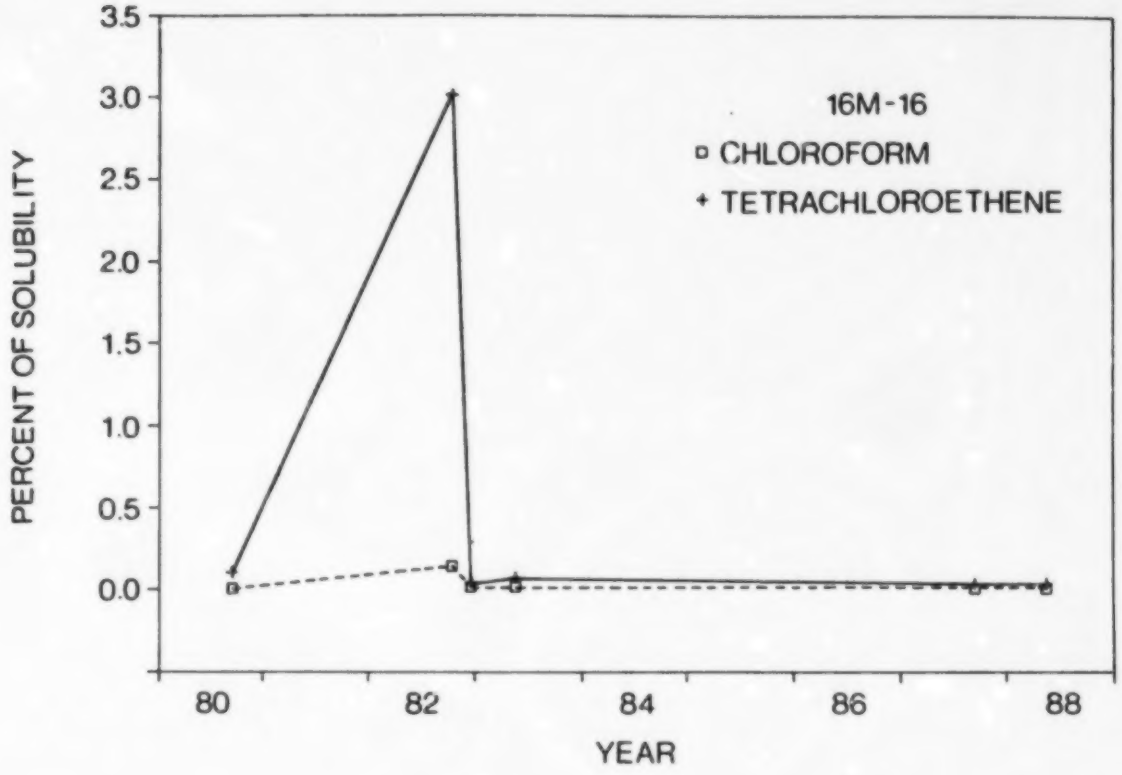
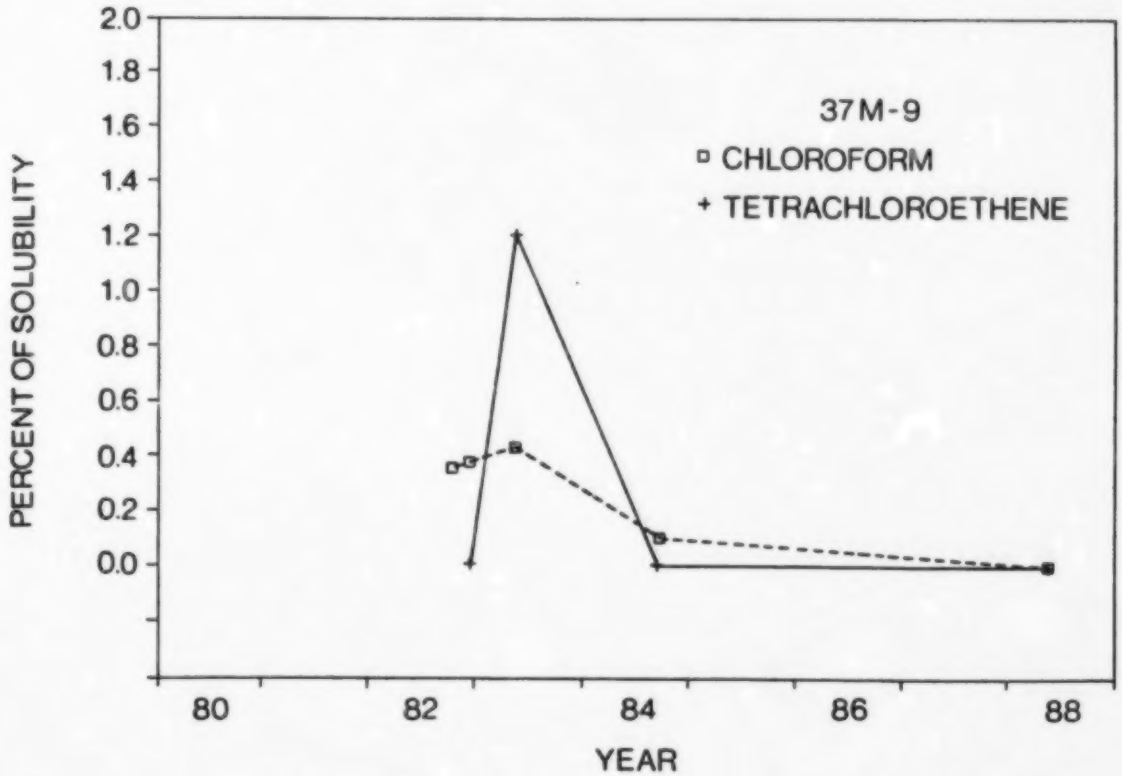

CONCENTRATION (ug/L)
0
4000
8000
12000
BENZENE AT 37M
DEPTH (m)
0
4
8
12
16
20
24
1982
1988
1984
0

Figure 23


16M-16
CHLOROFORM
TETRACHLOROETHENE
PERCENT OF SOLUBILITY
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
80
82
84
86
88
YEAR


37M-9
CHLOROFORM
TETRACHLOROETHENE
PERCENT OF SOLUBILITY
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
80
82
84
86
88
YEAR

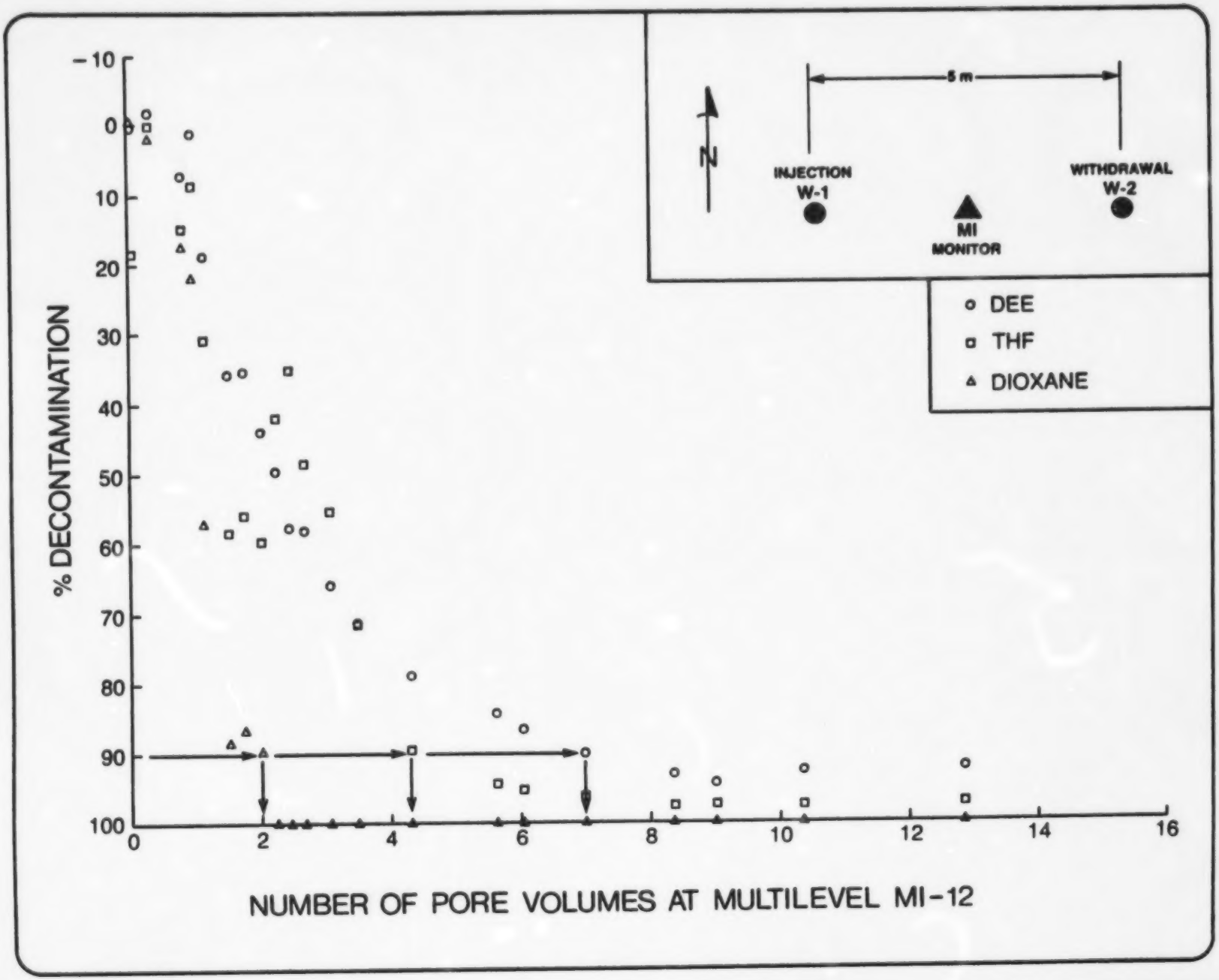
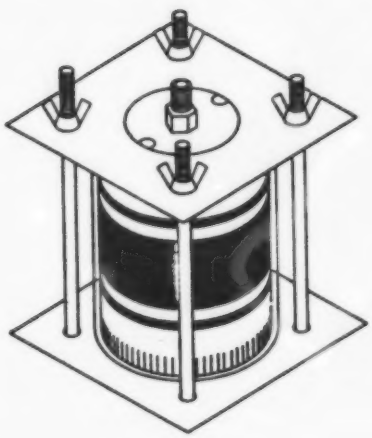
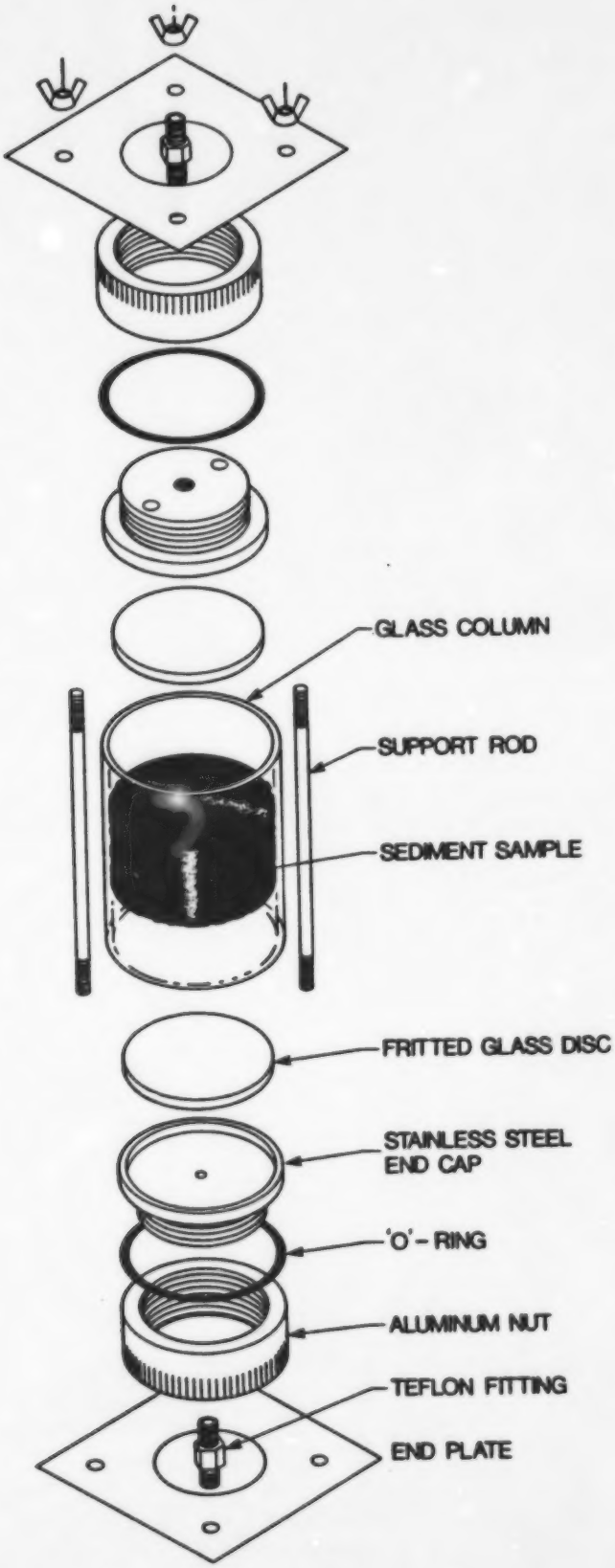

5 m
N
INJECTION
W-1
WITHDRAWAL
W-2
MI
MONITOR
DEE
THF
DIOXANE
% DECONTAMINATION
-10
0
10
20
30
40
50
60
70
80
90
100
0
2
4
6
8
10
12
14
16
NUMBER OF PORE VOLUMES AT MULTILEVEL MI-12

Figure 24

Figure 25




GLASS COLUMN
SUPPORT ROD
SEDIMENT SAMPLE
FRITTED GLASS DISC
STAINLESS STEEL
END CAP
'O' - RING
ALUMINUM NUT
TEFLON FITTING
END PLATE

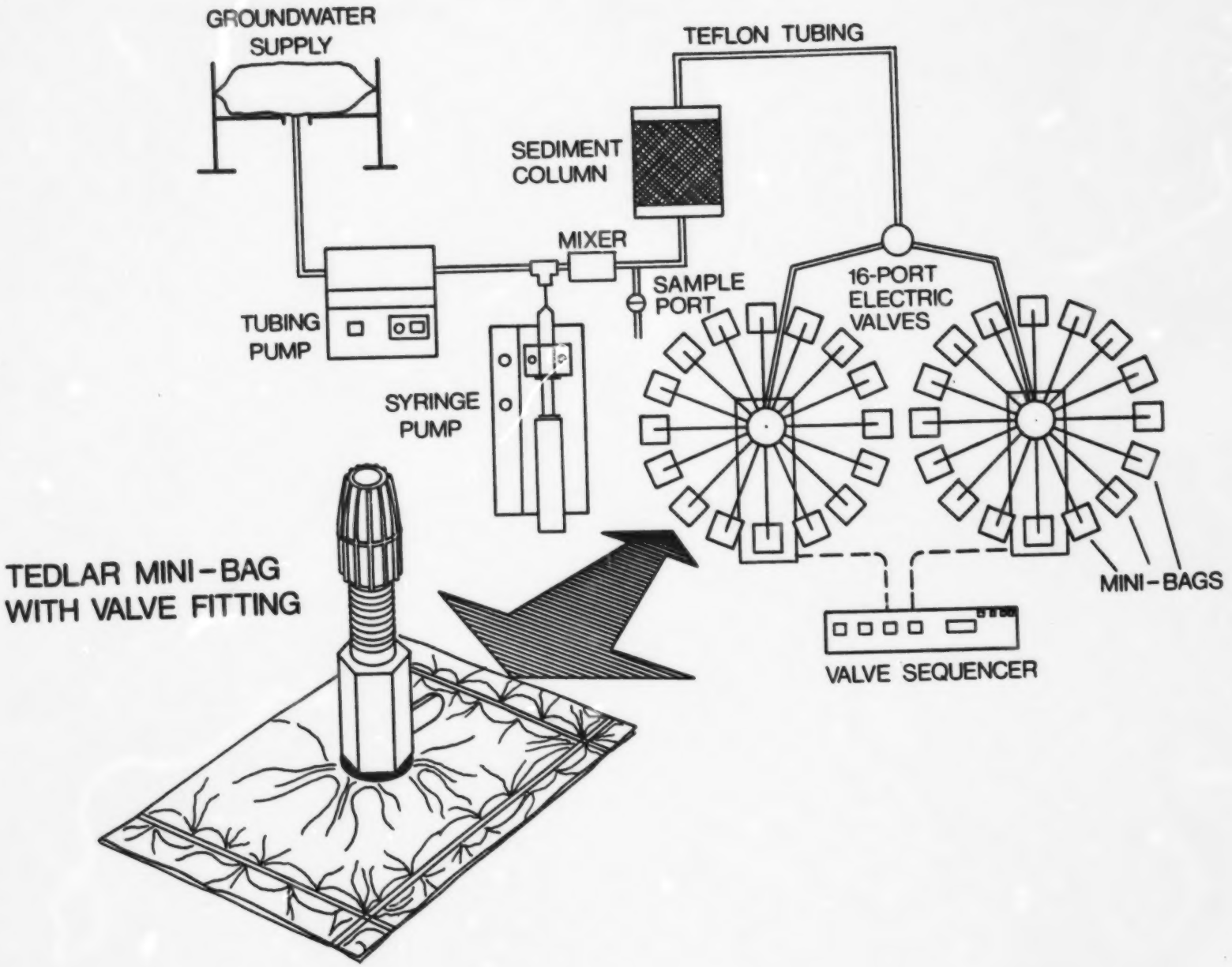

GROUNDWATER SUPPLY
TEFLON TUBING
SEDIMENT COLUMN
MIXER
SAMPLE PORT
16-PORT ELECTRIC VALVES
TUBING PUMP
SYRINGE PUMP
TEDLAR MINI-BAG WITH VALVE FITTING
MINI-BAGS
VALVE SEQUENCER

Figure 26

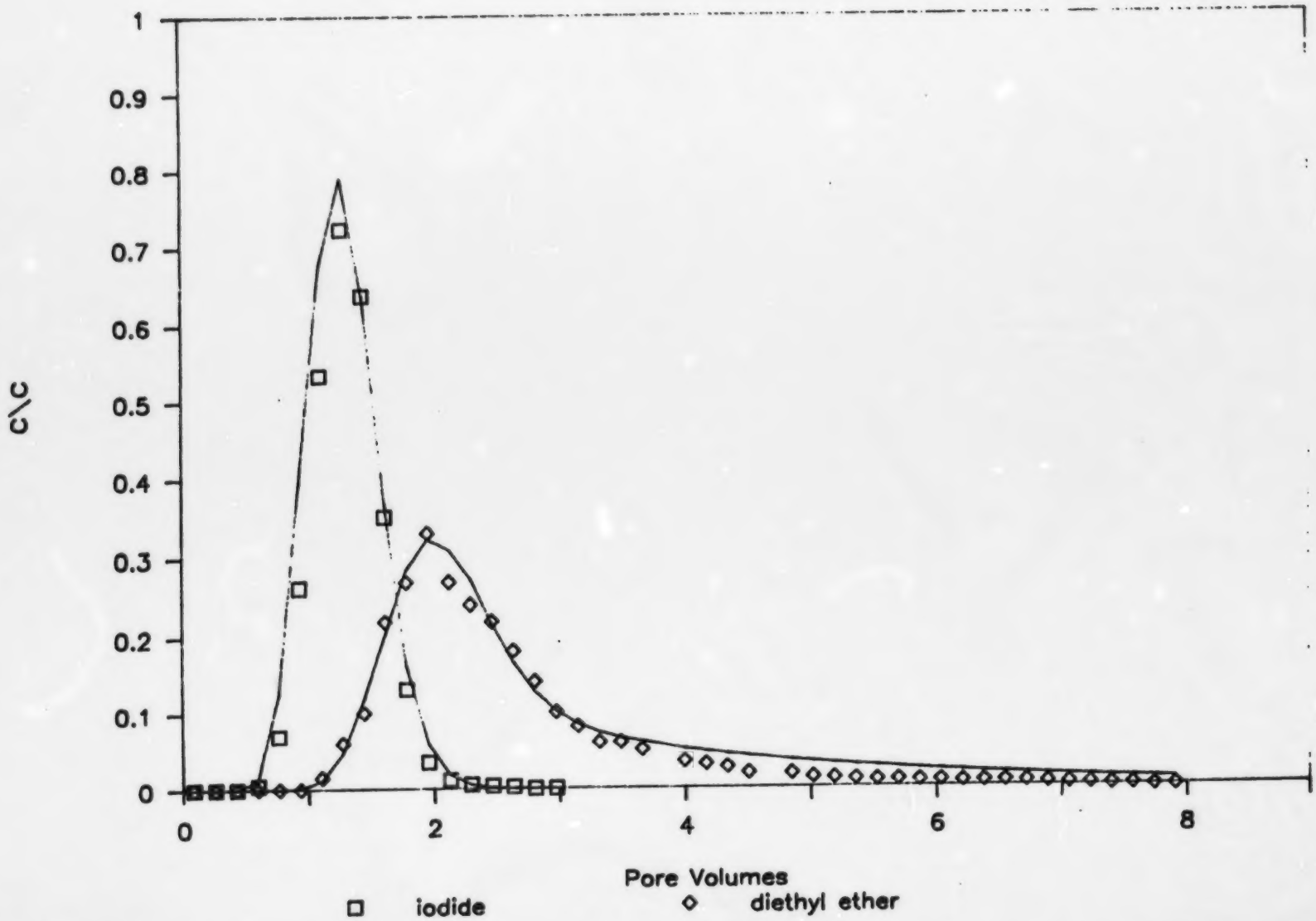

1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
c\c
0
2
4
6
8
Pore Volumes
iodide
diethyl ether

Figure 27

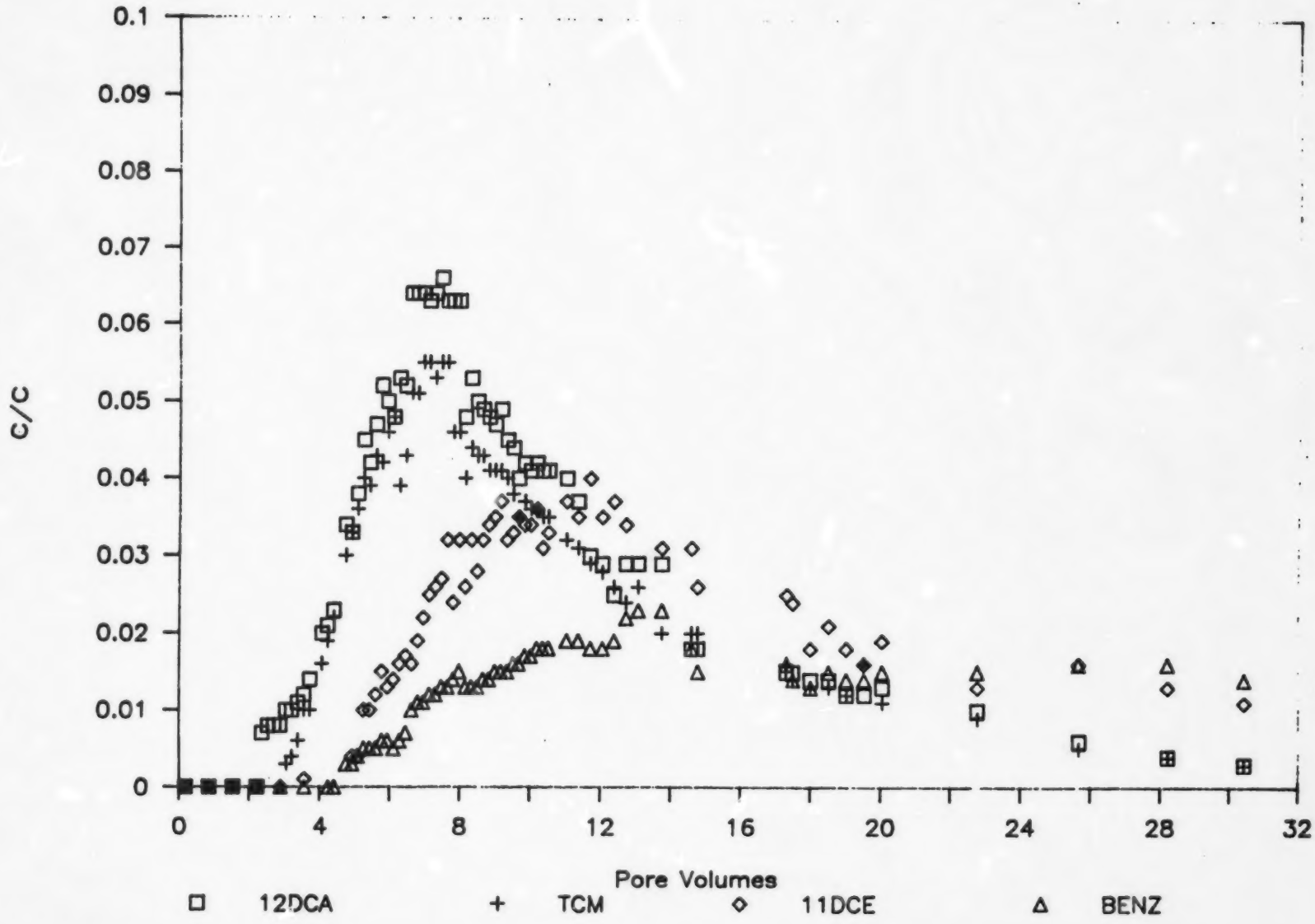

0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
C/C
0
4
8
12
16
20
24
28
32
Pore Volumes
12DCA
TCM
11DCE
BENZ

Figure 28

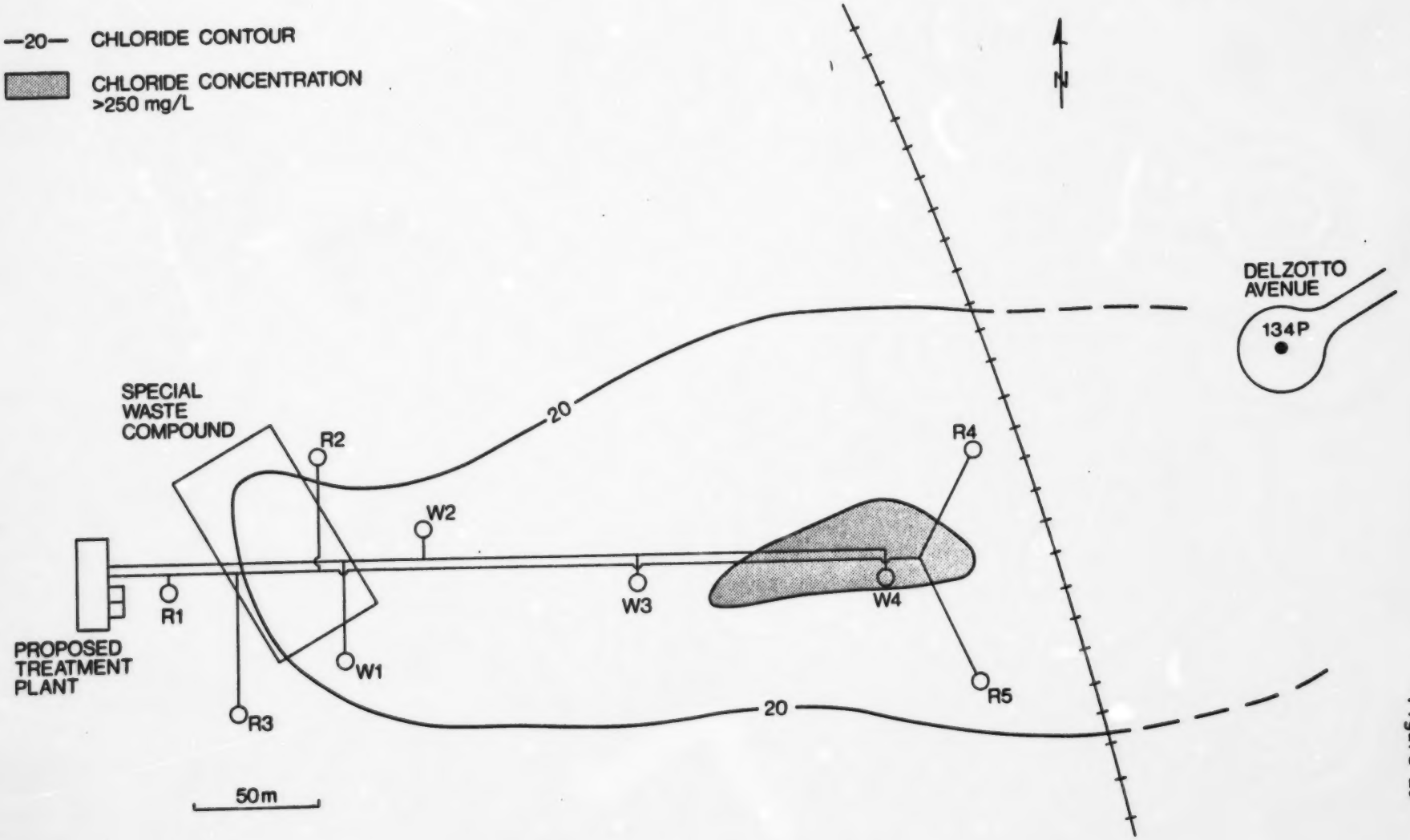
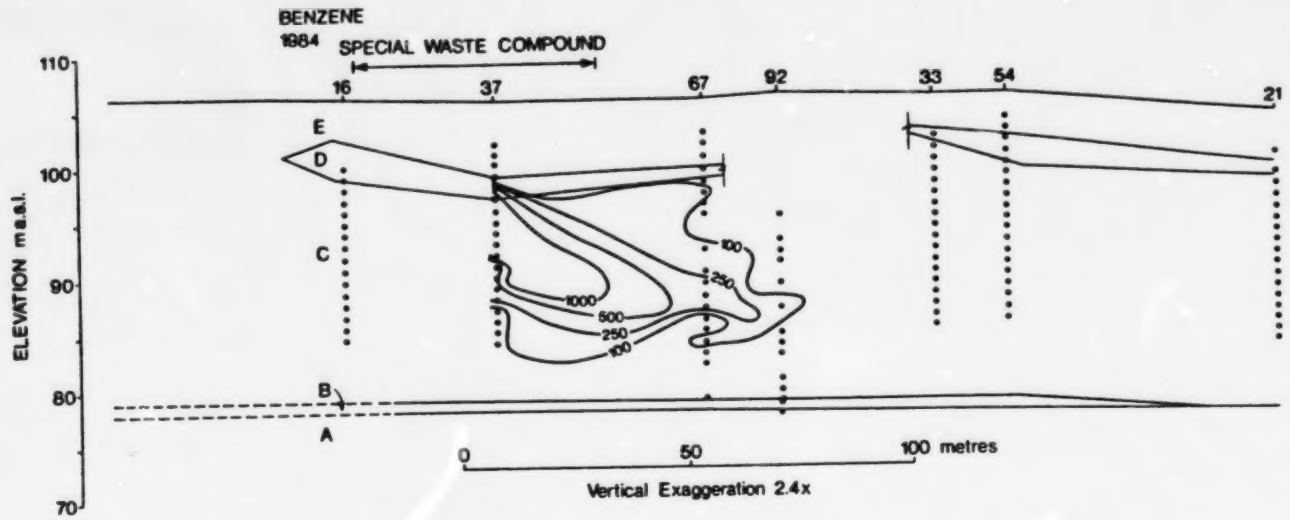
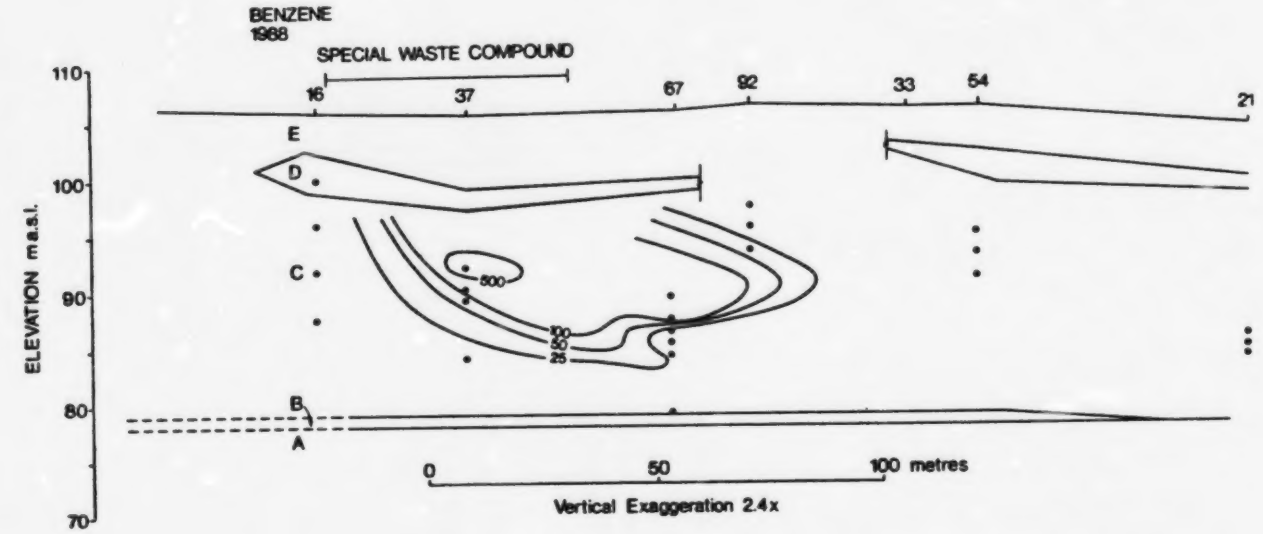
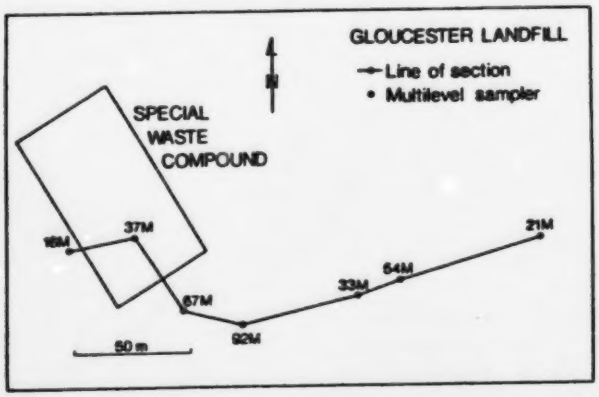

—20— CHLORIDE CONTOUR
CHLORIDE CONCENTRATION >250 mg/L
N
DELZOTTO AVENUE
134P
SPECIAL WASTE COMPOUND
R2
R4
W2
20
R1
W3
W4
PROPOSED TREATMENT PLANT
W1
R5
R3
20
50m

Figure 29

Figure 30







LEGEND

STRATIGRAPHIC UNIT

A Limestone; shaley limestone interlayers

B Till; coarse angular cobbles and boulders in sand and silt matrix

C Sub-aqueous outwash poorly sorted gravels, sands and silts

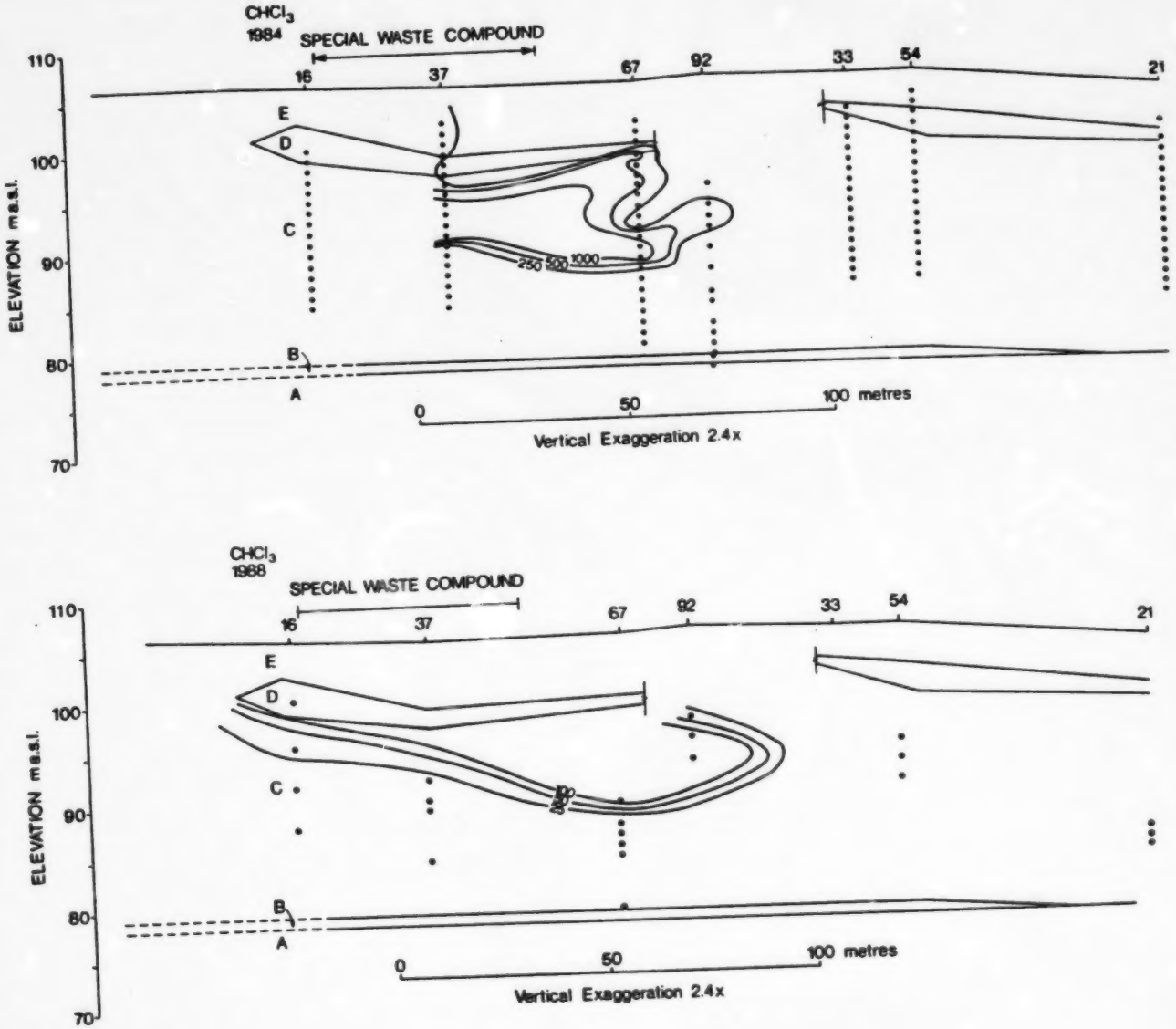
D Stratified clayey silt and silt

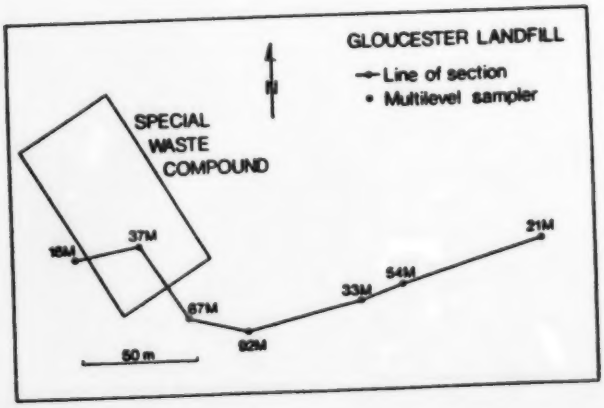
E Regressional sands and gravels

• Sample location

-100- Concentration contour in µg/L

Figure 31





LEGEND

STRATIGRAPHIC UNIT

A Limestone; shaley limestone interlayers

B Till; coarse angular cobbles and boulders in sand and silt matrix

C Sub-aqueous outwash poorly sorted gravels, sands and silts

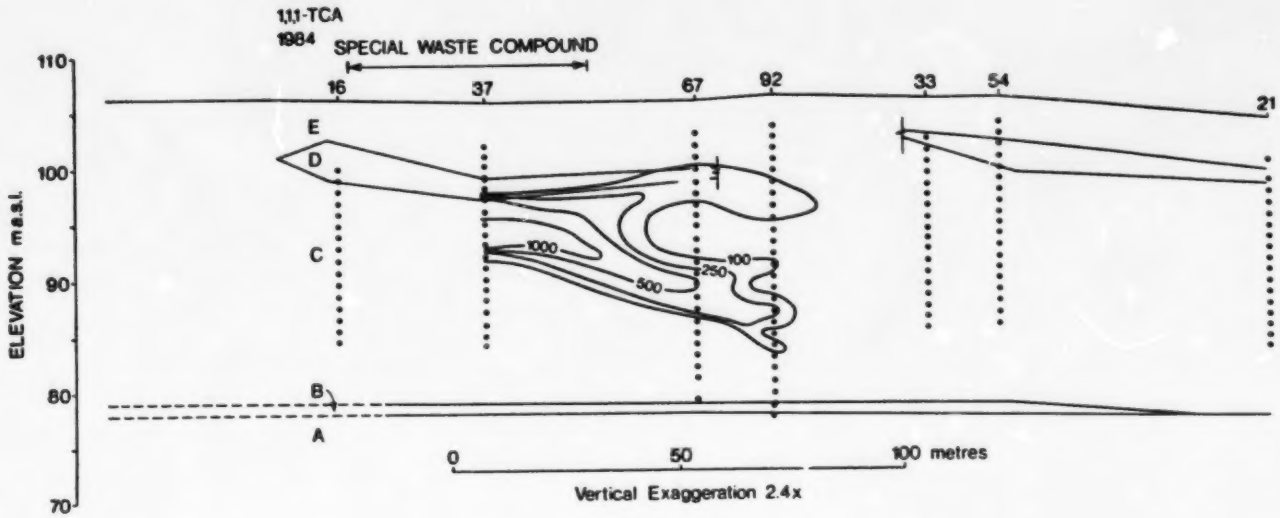
D Stratified clayey silt and silt

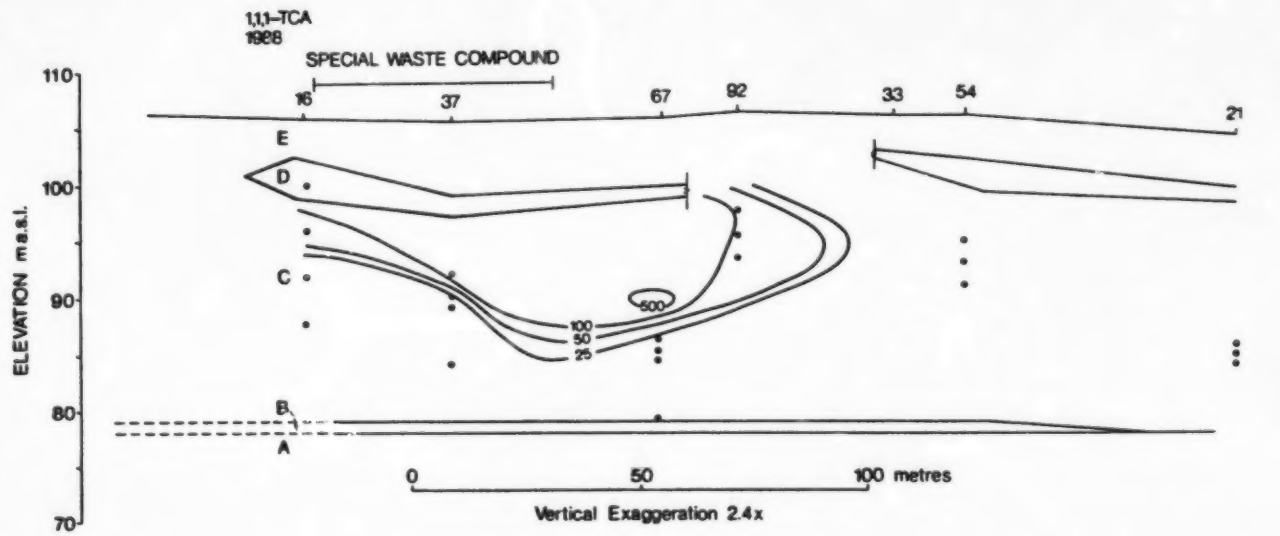
E Regressional sands and gravels

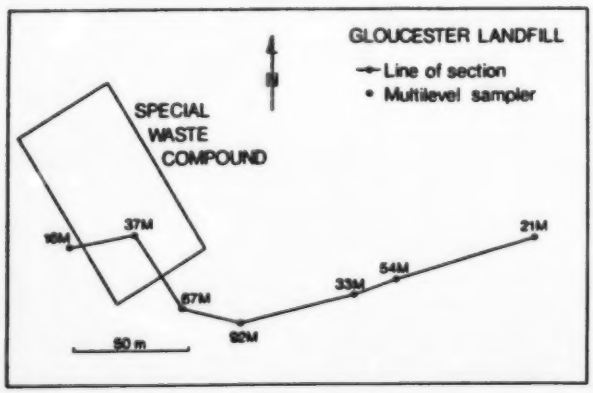
• Sample location

–100– Concentration contour in µg/L

Figure 32







LEGEND

STRATIGRAPHIC UNIT

A Limestone; shaley limestone interlayers

B Till; coarse angular cobbles and boulders in sand and silt matrix

C Sub-aqueous outwash poorly sorted gravels, sands and silts

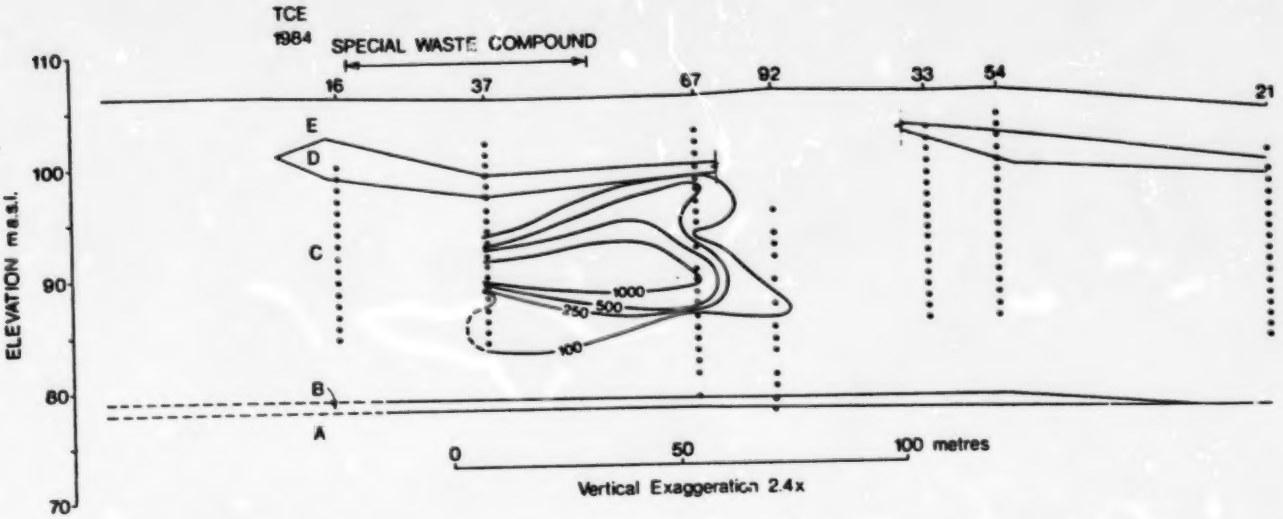
D Stratified clayey silt and silt

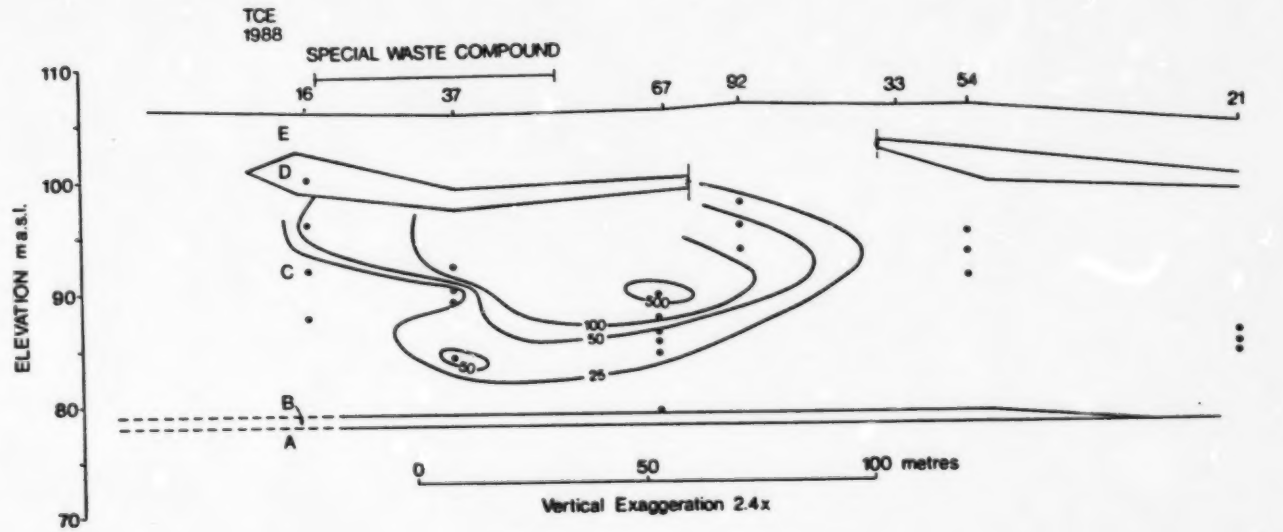
E Regressional sands and gravels

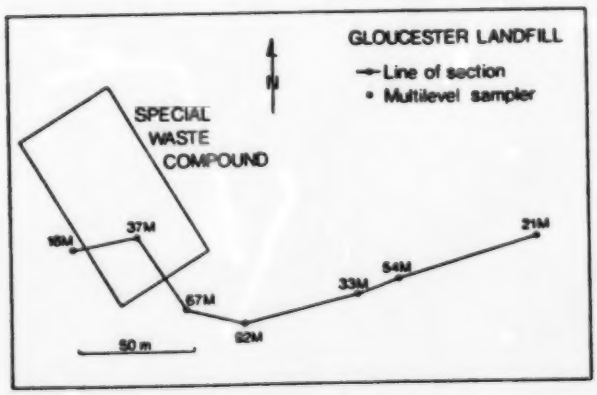
• Sample location

–100– Concentration contour in µg/L

Figure 33







LEGEND

STRATIGRAPHIC UNIT

A Limestone; shaley limestone interlayers

B Till; coarse angular cobbles and boulders in sand and silt matrix

C Sub-aqueous outwash poorly sorted gravels, sands and silts

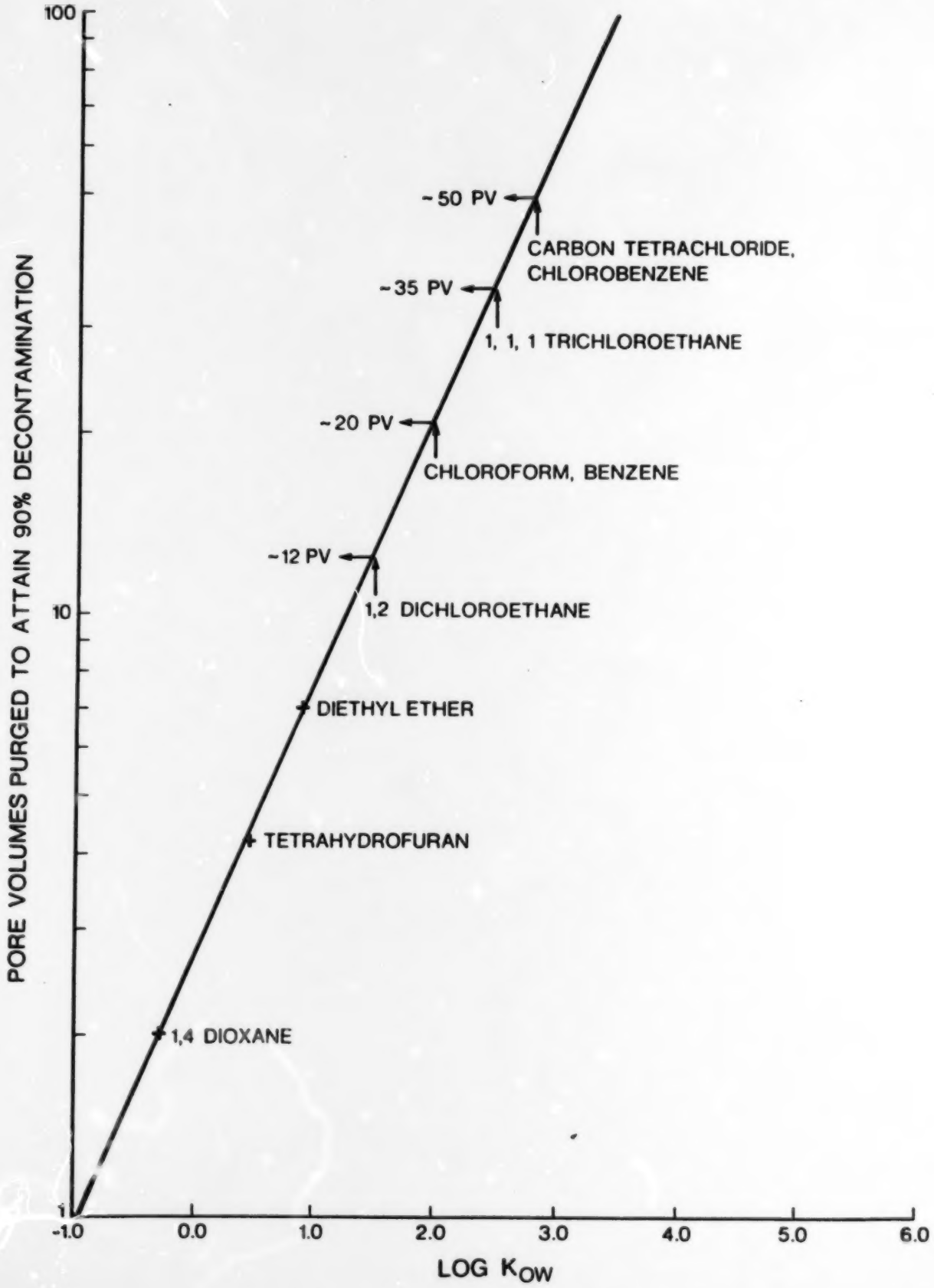
D Stratified clayey silt and silt

E Regressional sands and gravels

• Sample location

-100- Concentration contour in µg/L

Figure 34


100
10
1
PORE VOLUMES PURGED TO ATTAIN 90% DECONTAMINATION
~50 PV
CARBON TETRACHLORIDE,
CHLOROBENZENE
~35 PV
1, 1, 1 TRICHLOROETHANE
~20 PV
CHLOROFORM, BENZENE
~12 PV
1,2 DICHLOROETHANE
DIETHYL ETHER
TETRAHYDROFURAN
1,4 DIOXANE
-1.0
0.0
1.0
2.0
3.0
4.0
5.0
6.0
LOG K_{OW}